Federal/Provincial Research and
Monitoring Coordinating Committee (RMCC)

THE 1990 CANADIAN LONG-RANGE TRANSPORT OF AIR POLLUTANTS AND ACID DEPOSITION ASSESSMENT REPORT

Part 5

TERRESTRIAL EFFECTS

1990

ACID RAIN
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Canada

The 1990 Canadian long-range transport of air pollutants and acid deposition assessment report.

FEDERAL\PROVINCIAL

RESEARCH AND MONITORING

COORDINATING COMMITTEE (RMCC)

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TERRESTRIAL EFFECTS

Edited by

R. G. Pearson and K. E. Percy

1990

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Invited Preface

A great deal has been achieved over the past ten years. There has been a significant expansion of our scientific knowledge and our level of understanding of the environmental consequences of both acidic precipitation and the long-range transport of air pollution (LRTAP) on a regional scale. It became clear, however, that an integrated multidisciplinary approach was required to successfully address these regional scale environmental issues. Additionally, multijurisdictional cooperation was found to be absolutely essential. Despite this progress, there is still much which needs to be done. There is a desperate requirement for a more comprehensive and realistic understanding of the ambient natural environment and its inherent variability in all regions of Canada because this is the only way that one can interpret the significance of observed, measured, or predicted environmental changes in the future. This is particularly true with respect to the issue of the global atmosphere and climatic change which has emerged as the single most important environmental issue facing Canada and the world in the 1990's. The lessons gained from the research of the past ten years must be built upon and expanded so that our knowledge base is adequate to make the sound environmental management decisions which will allow for sustainable development, not only in the coming decade but in the 21st Century.

A. H. Legge

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5.1 SUMMARY

FORESTS

Acidic Wet Deposition and LRTAP

There is now general agreement that the current episode of maple decline in Canada is both more severe and more extensive than those which have occurred in the past. Although no evidence currently exists for a direct (foliar/airborne) or single component causal role for acidic deposition or ozone in any of the current tree declines in Eastern Canada, studies conducted in Ontario and Quebec indicate that acidic deposition and other long range transported air pollutants can not be ruled out as indirect (soil/nutrient) or contributing (pre-disposing) factors in sugar maple decline. The role and importance of these additional stresses on trees already under natural stress from climatic extremes (drought, temperature) as well as attack by insects and disease remains unknown but must be viewed with concern.

Symptoms of maple decline and tree mortality continued to increase in severity and extent during this assessment period. In many areas subject to acidic deposition and experiencing tree decline, soils have become deficient in several essential nutrients. Nutrition research has indicated that soil chemistry is an important factor in the decline syndrome and that in the short term, forest fertilization can provide an effective ameliorative treatment.

Dendrochronology data from two Ontario sugar maple studies also indicate that significant growth reductions have occurred in declining as well as outwardly healthy trees since the mid-1940s to mid-1950s in regions experiencing moderate to high levels of acidic deposition and ozone. However, these regions are also climatically different, thereby confounding direct causality relationships. In the case of white birch deterioration, research in New Brunswick has circumstantially linked repeated incidences of leaf browning and early senescence to exposure to acidic marine fog.

The lack of a continuous, historical Canadian forest condition data base has made it difficult to asses the severity and extent of current regional declines. With the exception of a few watershed-based projects in eastern Canada, there has been an inadequate program of intensive forest ecosystem monitoring from which predictions of the long term, subtle effects of pollutant loading on terrestrial and aquatic receptors can be made. Without such long term commitment, it has not been possible to define the multi-factor causality relationships or determine critical loading levels for soils and forests. Nor will it be possible to determine the effect of improvements in air quality on terrestrial systems.

Conclusions regarding the potential of acidic deposition have, of necessity, been based primarily on atmospheric monitoring data from major urban centres. A much more extensive and integrated monitoring effort based in the forest is required throughout all Canadian forest regions.

AGRICULTURE

Acidic Wet Deposition

No consistent foliar injury or crop yield responses which can be attributed to the direct effects of ambient levels of rain acidity occurring in rural areas of Canada have been documented under field conditions.

No interactive crop yield effects have been documented between acid rain and other LRTAP pollutants under simulated field conditions. However, only a few crop species have been evaluated under field conditions, and only a few of these experiments have been performed in Canada. Also, only a few cultivars of any one crop have been assessed. Most field-based research in the U.S. and Canada has now been terminated.

Ambient rainfall events in many rural areas of Canada now have lower pH's than experimentally derived acidity thresholds for the induction of foliar injury and reproductive effects on sensitive species.

Ozone

Foliar injuries to many sensitive crops have been documented in New Brunswick, Quebec, Ontario and British Columbia. Damage is often observed in rural areas where ozone hourly values are at or above the 82 ppb 1 hour Canadian objective.

Significant yield losses in sensitive crops have been experimentally documented in several studies conducted in Ontario and British Columbia. In Ontario, average annual yield losses in 19 species are estimated to range from 1 to 10% each year. Estimates of increased crop and ornamental productivity due to meeting the 1 hour ozone objective in Ontario range from $17 to $70 million annually, or up to about 4% of total annual sales of $1.9 billion.

No interactive effects between ozone and sulphur dioxide or acid rain/fog have been documented under simulated field conditions.

The rural ozone monitoring database in most areas of Canada is either non-existent or sparse, thereby limiting the assessment of potential crop impacts. Only a limited number of crop species have been evaluated under field conditions, with only a few of the studies being performed in Canada (Ontario and British Columbia). Even fewer cultivars of any one species have been assessed.

Current estimates of crop loss have been based on seasonal mean exposure:response data, primarily from the U.S. NCLAN program. However, it is now recognized that this exposure index may not adequately characterize crop response to season-long ozone exposure profiles. Activities have been initiated in both Canada and the U.S. to develop a more biologically relevant exposure index.

Other LRTAP Pollutants

No measurable yield response or foliar injury has been confirmed or would be expected due to ambient levels of PAN and NO_x in Canada. This conclusion, however, is based on a limited amount of high concentration, short term foliar response data, most of which was derived under controlled environment conditions. Ambient air monitoring of these compounds has been limited to major urban centres, with only isolated cases of rural coverage evident.

SOILS

Acidic Wet Deposition

First approximation maps describing the potential of soils and bedrock geology to reduce acidity are now available for Canada and indicate that approximately 46% of the land surface mapped to date is considered highly sensitive, 21% moderately so and 23% less sensitive. In terms of forest soil sensitivity, no data exist to assign critical or threshold loadings although research is now underway.

Although some baseline studies have been initiated to examine possible increases in soil acidification rates due to acidic deposition, there has been no coordinated attempt to assess these changes on a regional scale or to relate them to possible terrestrial impacts. Much more effort is required to develop sensitivity criteria most applicable to the acidification process in representative Canadian soils and to relate these factors to critical loading.

TERRESTRIAL WILDLIFE

Acidic Wet Deposition and LRTAP

Defoliation in declining maple forests of Quebec has been shown to reduce bird populations in tree canopies and to increase populations of some species in ground cover vegetation.

Lichens and some other food sources of grazing ungulates have been shown to accumulate metals when grown on poorly bufferred soils in acid-sensitive areas of Ontario. These food chain effects have resulted in accumulation of metals in tissues of grazing animals such as moose and deer in certain areas of Ontario. Limited availability of ungulate food sources has resulted in arctic and sub-arctic areas where mosses and lichens have been affected by LRTAP pollutants.

Research on the indirect effects of acidic deposition on wildlife has only recently commenced in Canada. Causality relationships with pollutants will be difficult to document due to their indirect nature and association with other terrestrial impacts.

5.2 INTRODUCTION

The purpose of this Canadian assessment document was to review the past four years of scientific effort to provide a basis for future decisions on continued research, monitoring, and control strategies and to facilitate evaluations of the efficacy of Canadian and international abatement programs. In order to achieve this task, the Terrestrial Effects Sub-Group (TESG) of the Federal-Provincial Research Monitoring and Coordinating Committee (RMCC) posed six questions which have served to focus this report. The authors who were chosen to address the various components of the LRTAP issue were selected on the basis of their research and monitoring experience in Canada. The lead authors were directed to coordinate the review of the current state of knowledge in their respective program areas, with emphasis on progress that has been made in Canada since the last major assessment document in 1986. They were further directed that this was not to be a comprehensive global literature review, as many excellent summaries of the open literature have already been published. To assist the authors in the compilation of their respective chapters the TESG initiated a Canada-wide registry of current and past LRTAP research and monitoring programs, complete with a brief progress summary and contact scientist for each project. Although this document was not printed in final form until after the completion of the assessment process, it was distributed in draft format to all authors to assist them in contacting other Canadian scientists who have and continue to contribute to the advancement of the science.

The questions for the 1990 assessment are summarized below:

1.0 **What levels of acidic wet deposition, ozone and associated pollutants can be tolerated without significantly affecting the terrestrial environment?**

2.0 **What improvements due to Canadian and/or United States emission reduction scenarios, can be expected on the terrestrial environment?**

3.0 **What is the risk that acidic wet deposition, ozone and associated pollutants will damage the forest?**

3.1 What forest areas are most likely to be affected?

3.2 What is the likely magnitude of the effect?

3.3 What is the extent of economic losses in forestry?

4.0 **What role do acidic wet deposition, ozone and associated pollutants play in causing hardwood decline?**

4.1 What are the economic consequences of sugar maple decline?

4.2 What management methods can mitigate sugar maple decline?

5.0 **What is the risk of acidic wet deposition, ozone and associated pollutants reducing agricultural production?**

5.1 What crops in what areas are most likely to be affected?

5.2 What is the likely magnitude of the effect?

5.3 What is the extent of economic losses in agriculture?

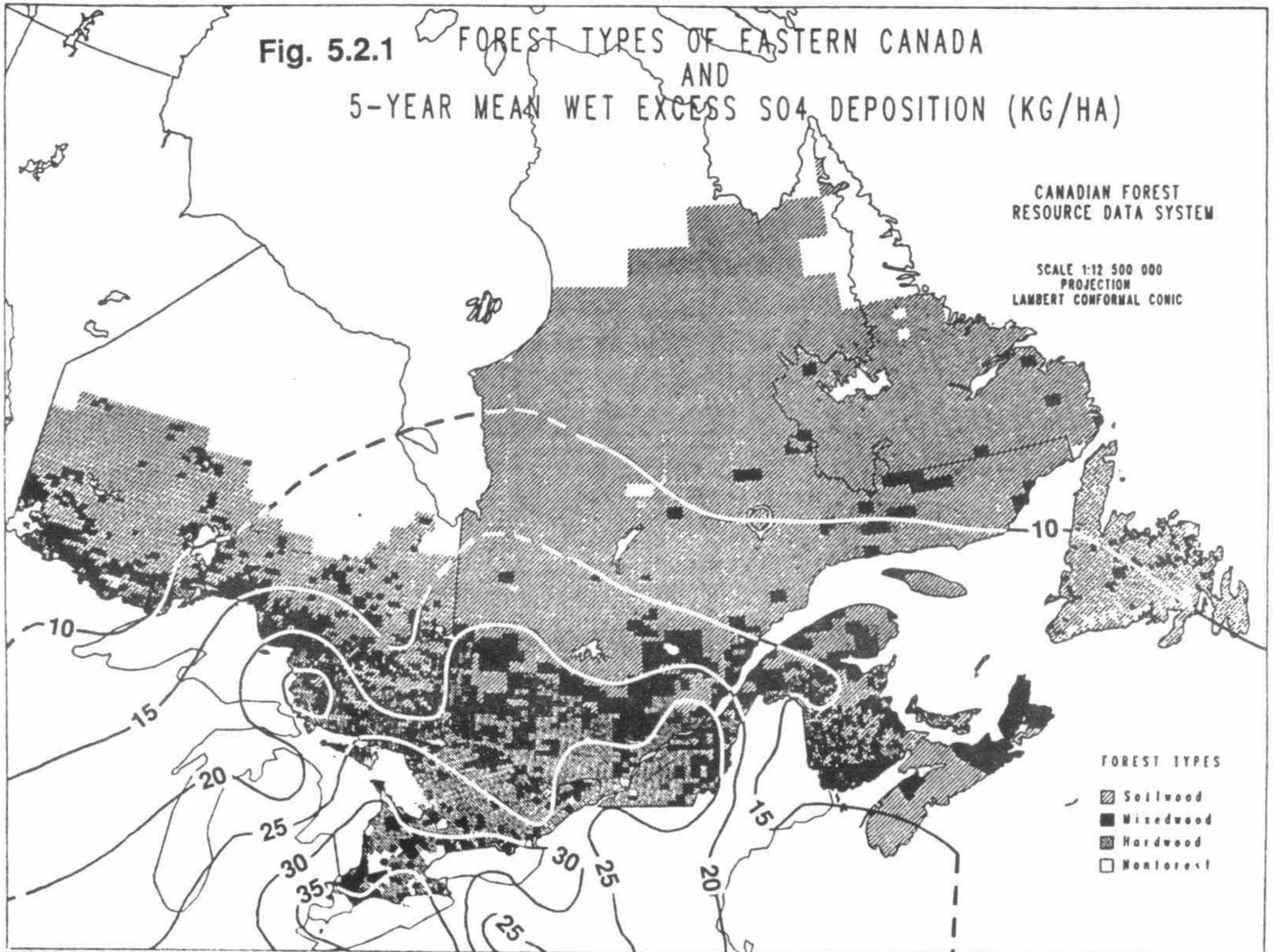
6.0 **What is the risk of acidic wet deposition, ozone and associated pollutants affecting terrestrial wildlife?**

5.2.1 RESOURCES AT RISK

There are 10 major forest-vegetation regions in Canada (Rowe, 1972). In eastern Canada, purely deciduous forests are restricted to southern Ontario. North of this in Ontario and throughout southern Quebec is the Great Lakes-St. Lawrence mixedwood forest, and north of this again is the Boreal, primarily coniferous forest. There is no scientific basis to relate the existing aquatic threshold (20kg/ha/yr) for wet deposited acidic sulphate to terrestrial systems. However, if it were utilized as an indicator for potential impacts, then it is primarily the hardwood and the mixedwood forests (approximately 15 million hectares) that are exposed to potentially significant regional sulphate deposition (**Figure 5.2.1** -Addison, 1989). The boreal forest, from which the bulk of Canada's forest products arise, is not exposed to sulphate levels in excess of 20 kg/ha/yr.

Nitrate deposition patterns (**Figure 5.2.2**) follow sulphate loading closely, supporting

Fig. 5.2.1 FOREST TYPES OF EASTERN CANADA AND 5-YEAR MEAN WET EXCESS SO4 DEPOSITION (KG/HA)



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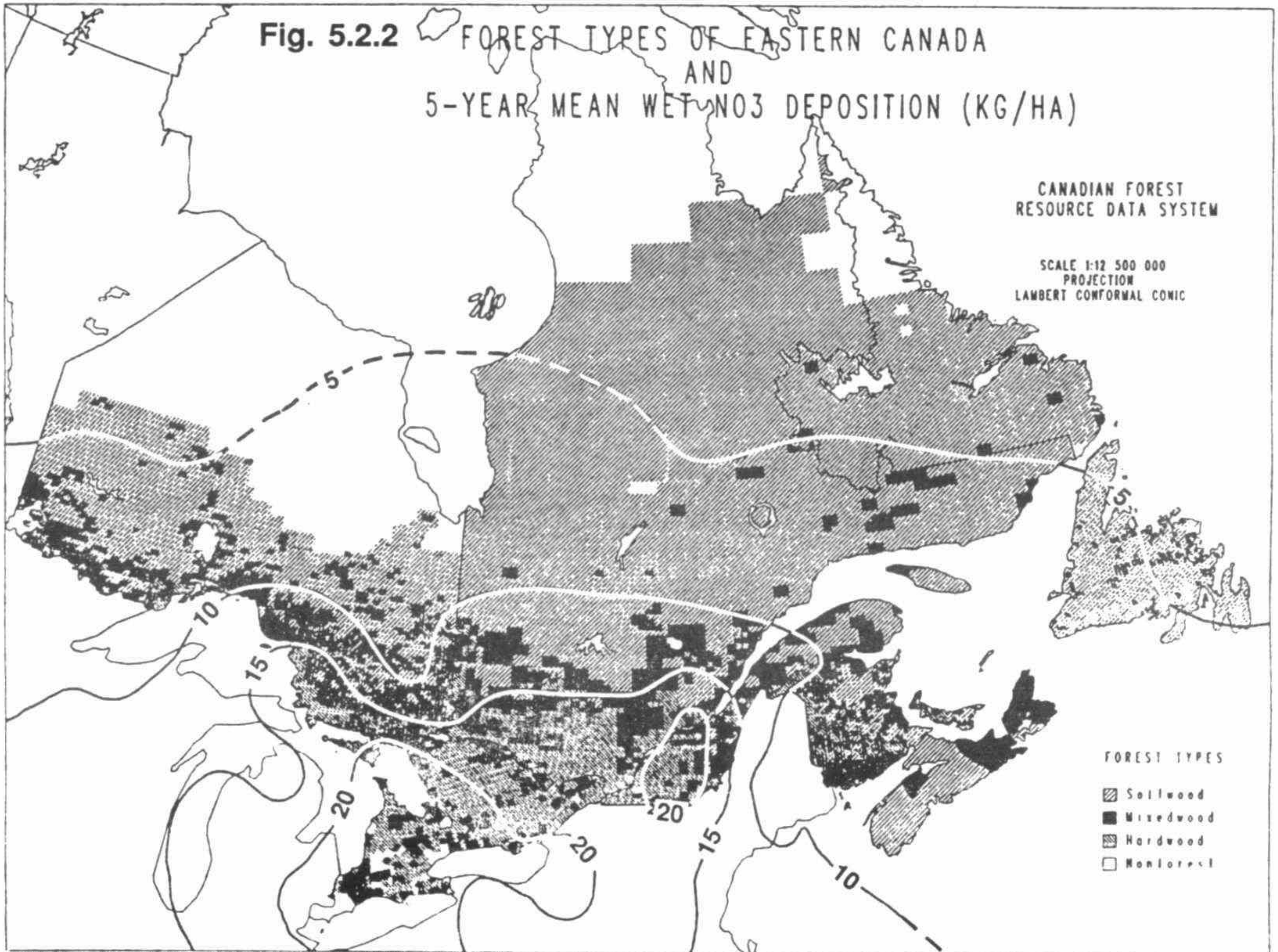


Fig. 5.2.2 FOREST TYPES OF EASTERN CANADA AND 5-YEAR MEAN WET NO3 DEPOSITION (KG/HA)

PRODUCED AT THE PETAWAWA NATIONAL FORESTRY INSTITUTE, FEBRUARY 1989

the hypothesis that these two major acidic contributors represent the distribution of regional, acid producing air pollution as a whole. Although there is evidence that ozone air pollution tends to follow a similar pattern in parts of eastern Canada, there is inadequate ozone monitoring throughout all of Canada to verify this trend in other regions.

In western Canada, the pollution monitoring network is not sufficiently dense to permit this type of mapping of pollution deposition. It is evident, however, that while acidic deposition is much reduced regionally, there is significant deposition in the lower mainland of British Columbia. In this area there are also elevated ozone levels; however, monitoring networks are not adequate to assess the geographical distribution of this pollutant. In the absence of adequate data, it is estimated that 2-3 million hectares of forested land may be exposed to high pollution levels in British Columbia, close to the Vancouver population centre.

In Canada therefore, the forests most exposed to regional air pollution are those close to populated areas. These forests are some of both the most productive in the country and the most heavily utilized. Typical uses include recreation, tourism, wildlife habitat, aesthetics and economics (woodlots, maple syrup, quality hardwood and softwood lumber). It is concern over these forests that drives the air pollution-forest issue for Canadians.

From an agricultural perspective, potential impacts of LRTAP give reason for greater concern based on deposition patterns which are highest in some of the most productive agricultural areas of Canada. Of the total 33 million hectares of Canadian crop land, approximately 5 million (15%) are located in the most heavily impacted areas of eastern Canada and British Columbia. However, from a total sales perspective, these areas account for over 3 billion (33%) of the total 8.9 billion dollars in annual agricultural sales (Statistics Canada, 1986, 1989). Unlike softwood forestry operations, which, for the most part, are located remote from the influence of the heaviest pollutant loading, crop production is generally located near the more heavily populated and pollutant stressed urbanized areas.

5.3 TERRESTRIAL MONITORING

5.3.1 EXISTING SYSTEMS

5.3.1.1 INTRODUCTION

For the purpose of the present discussion, 'terrestrial monitoring' is considered as the direct measurement of potential changes in or responses of receptor ecosystems. The aim of direct monitoring is to establish baselines representing the current status of terrestrial ecosystems and the various components thereof. From this Baseline, it will then be possible to detect both anthropogenic and natural change over time.

In Canada, terrestrial monitoring activities have largely, though not exclusively, been concerned with forest ecosystems and the soils which support them. This is because of the economic importance of forestry and the fact that extensive areas of hardwood forest are found on sensitive terrain which is exposed to high levels of air pollution.

Timber growth has been monitored by a variety of means for decades or even centuries in all jurisdictions where forest management is practised. Such monitoring has usually taken the form of inventories where an entire resource or portion thereof is surveyed and changes from the previous inventory determined. In some cases permanent plots have been established where growth is calculated from measurements repeated over time. Factors such as insect infestations and disease occurrences have likewise been monitored in most jurisdictions for many years. Thus, it was logical to adapt survey and permanent plot techniques to the current air pollution problem.

During the late 1970s and early 1980s, severe and unexplained damages on a number of commercially important forest species including silver fir, Norway spruce, Scotch pine and beech were identified in the forests of central and western Europe, especially FR Germany, but also in the German DR, Czechoslovakia, France, Switzerland and elsewhere (Bucher and Bucher-Wallin, 1989).

In Europe, twenty-nine countries (or in a few cases regions within countries) plus the European Economic Community have conducted coordinated forest damage surveys under the aegis of the United Nations (UN) Economic Commission for Europe (ECE) since 1985 using a common methodology developed in the FRG (Anonymous, 1986). The surveys have taken the form of a line-plot survey on a minimum 16 km x 16 km grid. In some countries, such as FR Germany, more intensive samplings were made (Anonymous, 1987; Anonymous, 1988; Anonymous, 1989). Damage was assessed in terms of 5 classes (none, slight, moderate, severe, dead), derived from a combination of 5 defoliation classes modified by 3 discolouration classes. In addition to the damage survey, plans are being laid for second and third levels of monitoring. The second level will involve a network of permanent (as opposed to temporary) sample plots; the third, a network of ecosystem studies.

While a signatory to the UN Convention on Long-range Transboundary Air Pollution, Canada (as well as the USA) has participated mainly as an observer in the aforementioned 'International Cooperative Programme on Assessment and Monitoring of Air Pollution Effects on Forests in the ECE Region'. The extent of Canada's forest area (436 million ha) as opposed to 7.4 million for FR Germany or 1.2 million for Switzerland, has precluded ground-survey effort at the same intensity as in Europe.

5.3.1.2 SURVEYS

The cornerstone of the Canadian detection system is the Forest Insect and Disease Survey (FIDS) of Forestry Canada. Originating in 1936 as the Forest Insect Survey, and expanding in 1951 to become the FIDS, this organization relies on permanent field rangers experienced in recognizing forest damage.

Provincial forest and/or environment departments have conducted surveys, particularly in response to specific problems such as maple decline observed in parts of Quebec and southern Ontario during the early 1980s. In Quebec, the Service de Protection Contre les Insectes et les Maladies of the Ministère de l'Energie et des Ressources (MERQ) undertook the aerial mapping of affected forests in 1983. Stands were characterized as healthy, or in light, moderate or heavy stages of decline, based on foliage loss. By 1985, about 25,110 km^2 had been mapped (Carrier, 1986). In Ontario, the Ministry of the Environment (MOE) established 110 permanent sample plots in 1986, all within the range of sugar maple in southern Ontario, under the Hardwood Decline in Ontario Survey. These were re-assessed in 1987 and again in 1989 in terms of tree condition. Samples of foliage and soil also were collected for analysis (McIlveen and Linzon, 1988).

5.3.1.3 PERMANENT PLOT NETWORKS

An Acid Rain National Early Warning System (ARNEWS) was established in 1984 by the FIDS of Forestry Canada. This consists of a network of 0.04 ha permanent sample plots (currently numbering 106) distributed throughout all major forest types and all provinces of Canada. Plots are assessed several times annually for specific 'acid rain symptoms', as well as insect and disease conditions. In addition, they are assessed annually, for tree condition and tree mortality, and every five years for tree growth and foliage and soil chemistry (Magasi, 1988). The basic approach of the ARNEWS has been to determine the state of a forest's health in general, rather than to concentrate on specific pollutant responses. If observed damage cannot be explained by reference to known agents, further investigation is undertaken. In fact, "The ultimate outcome of the monitoring system if unexplained damage is detected is a research project on possible causes" (Addison, 1988). In Manitoba, the Department of Environment, under the Terrestrial Monitoring in Manitoba program, is maintaining a network of eight permanent sites, four in conjunction with the ARNEWS.

In addition to the decline survey noted above, the Ontario MOE established in 1984, and is currently maintaining, a network of eleven permanent sample plots (eight in the Muskoka-Parry Sound-Algonquin area, two in the Peterborough area, with a control site near Thunder Bay, Ontario) under the Sugar Maple Decline in Ontario Study. These plots are assessed annually for tree condition and growth, as well as for foliar and soil chemistry (McIlveen and Linzon, 1988).

Also, in relation the maple decline problem, the MERQ, under the Deperissement et Mortalite dans les Erablieres project and following from the aerial survey work noted above, has established a network of 256 permanent sample plots throughout the natural range of sugar maple in Quebec. Following initial dendrometric, pathological, entomological and ecological characterization, these plots are assessed annually in terms of sugar maple decline symptoms. A subset of 26 of these is more intensively-monitored under the Impacts des Differents Stress Environnementaux sur la Dynamique et la Productivite des Ecosystemes Forestieres Project. These plots are paired with Ministère de l'Environnement du Quebec air pollution monitoring stations. In addition, seven of these plots are part of the North American Sugar Maple Project. The latter is a joint Forestry Canada/USDA Forest Service (with the cooperation of several provincial and state forest departments) sample plot network across the range of sugar maple, consisting of 60, 0.04 ha plots in Canada and 106 in the USA. These plots are in pairs, consisting of a (tapped) sugarbush and (untapped) maple woodlot control. These are assessed annually (Millers and Lachance, 1989).

Observations were first made on white birch deterioration along the Bay of Fundy, New Brunswick, by the FIDS in 1979. Currently 13 permanent sample plots are maintained under the White Birch Deterioration Project.

Specifically oriented to soil, as opposed to forest health monitoring, Alberta Environment, since 1981, has maintained a network of eight permanent sites to be sampled every four years under The Long-Term Soil Acidification in Alberta Project. In Ontario, the MOE, beginning in 1980, commenced its Soil Baseline study. Initially 400 sites were sampled between 1980 and 1981. A subset of 100 was re-sampled in 1987 and analyzed for 25 parameters including pH, cation exchange capacity, base saturation, organic matter, iron, aluminum, metals, sulphur and nitrogen.

A tree foliage analogue of this in Ontario has been the Baseline Tree Foliar Chemistry Survey, based on a network of 12 permanent sites, all in southern Ontario, maintained by the MOE. These were sampled in 1980-81 and again in 1987. The MOE also maintains other sites in relation to some of its other LRTAP projects.

5.3.1.4 WILDLIFE MONITORING

Forest decline has the potential to significantly impact native wildlife. Invertebrate communities that exist within forests and their soils also can be directly or indirectly altered by atmospheric pollutants including sulphur, acids and ozone (Gilbert, 1971; Hagvar and Amundsen, 1981; Coleman and Jones, 1988).

Several jurisdictions, including Manitoba (Wotton and McEachern, 1988), Ontario (Glooschencho et al., 1988), Quebec (Crete et al., 1987), New Brunswick (Ecobichon et al., 1988), Nova Scotia (Sullivan, 1989) and Newfoundland (Brazil, 1989) have monitoring programs to determine the content of trace metals in organs of herbivores, including moose, deer, bear (Redmond, 1989), caribou (Brazil, 1989)

and snowshoe hare (Brazil, 1989).

The forest bird monitoring program (FBMP), begun in 1987 by the Canadian Wildlife Service, collects standardized bird survey data in large forest stands across Ontario Sixty-two sites were surveyed in 1987 with the aim of eventually establishing 150 sites. A link to ongoing forest health monitoring programs would enable a better understanding of impacts of acid rain and forest dieback on bird populations. Other long-term bird surveys may also be used to interpret acid rain effects (DesGranges, 1987).

The results of these investigations are presented in Section 5.4.3.

5.3.1.5 WATERSHED STUDIES

Five ecosystem or 'watershed' studies in eastern Canada are identified with the LRTAP program. These are, from west to east (1) Experimental Lakes Area (ELA), northwestern Ontario, (2) Turkey Lakes Watershed, Algoma District, Ontario, (3) Dorset, central Ontario (4) Montmorency Forest, Quebec, (5) Kejimkujik National Park, Nova Scotia. Major terrestrial programs involving environmental monitoring have been undertaken at the last four. A significant program is still underway at Turkey Lakes Watershed. While they have varied with the objectives of the particular research project, monitoring aspects have generally (though not always) included records of tree growth, precipitation, throughfall, stemflow, and litterfall amount and chemistry, floor and mineral soil percolate flow and chemistry, and stream flow amount and chemistry. The work is frequently integrated with the work of other agencies. At the Turkey Lake Watershed, it has involved Forestry Canada, Fisheries and Oceans Canada and Environment Canada (principally, National Water Research Institute and Atmospheric Environment Service) and the cooperation of the Ontario Ministry of Natural Resources (Jeffries et al., 1988a).

5.3.1.6 SUMMARY

Three levels of terrestrial monitoring may be identified, viz. surveys (aerial and ground), permanent plots, and ecosystem (watershed) studies. Ideally these should relate to one another in such a way that information can be transferred from one level to the next. At the same time, forest monitoring should be integrated as much as possible with other environmental interests, particularly air, water and wildlife, so that information can be transferred from one sector to another.

A comprehensive damage survey, such as has been carried out in Europe over the past several years, would be impractical in Canada because of the diversity and extent of the Canadian forest. Limited surveys have been carried out by the MERQ in Quebec and the MOE in Ontario as well as elsewhere. Overall, surveillance is provided by the FIDS. Because a number of agencies are involved, communication must be maintained and efforts must continue towards standardization of

assessment techniques.

The ARNEWS approach, which involves determining the overall state of a forest's health, rather than focusing on specific pollutant responses, is well adapted to the Canadian situation. It has the advantage that currently-unenvisaged forms of damage could be detected. The current number of ARNEWS plots is small and should be expanded considerably. As much as possible, plots should be located to take advantage of deposition networks and other studies.

Several watershed studies have been conducted in Canada during the past decade. A small network of well-chosen ecosystem (watershed) study sites should be established representing the major forest types at least in and around the major pollutant deposition zones. In addition to the basic research carried out at such sites and the monitoring value of following ecosystems in their entirety, such sites should serve as cross-over points where various networks co-locate.

5.3.2 SOIL SENSITIVITY MAPPING

5.3.2.1 INTRODUCTION

Assessments of terrestrial sensitivities to acidic deposition were initiated in Canada in response to the Canada-U.S. Memorandum of intent (MOI) agreement on transboundary air pollution in 1980. The related mapping effort had the following objectives:

- to review and compile existing soil and geology information useful for assessing environmental degradation,
- to identify areas where high acidic deposition and high sensitivity correspond and, therefore, more intensive investigations are required,
- to provide a framework for identifying data or information gaps,
- to provide a logical base for future long-term monitoring.

The following summarizes the approach taken in Eastern and Western Canada (including Northern Canada), and outlines a national perspective.

5.3.2.2 APPROACH

Terrestrial sensitivity is an interpretation which relates anticipated responses of soils and geological materials to acidic deposition. Such responses pertain to a decrease

in pH, base cation loss, sulphate and nitrate retention, aluminum mobilization, solubilization of heavy metals, and enhanced silicate and carbonate weathering. For convenience, and in the absence of dose-response information, sensitivity assessments have been based on rating individual mapping units of the land (soils, surficial deposits, bedrock). Rating schemes may be simple or complex, with different keys being developed for different interpretations or objectives. Simple classifying schemes of the type shown in **Table 5.3.1** are most conducive for qualitative interpretations of existing soil and bedrock maps. Typically, map units are assigned to three sensitivity classes (low, moderate, high).

Thus far, most attention has been focused on rating the long-term capacity of soils and geologic materials within watersheds to buffer and modify the impact of acidic input on drainage water, streams and lakes. This has given rise to maps showing the potential of soils and geology to reduce acidity. Also in use are soil sensitivity maps, produced by rating the susceptibility of soils to chemical and physical changes which could affect plant growth.

Process models in combination with appropriate classification schemes are required for quantitative interpretations. One example is the method for the sensitivity mapping of agricultural land developed by Coote et al. (1989). These authors rate agricultural soils by estimating the anticipated loss of bases as affected by acidic deposition and by N fertilization. The loss of bases is estimated from current and expected acid loadings (by region), and from the capacity of each soil type to buffer the effective acid stress.

5.3.2.2.1 Eastern Canada

The initial assessment was done by Shilts et al., (1981) who produced terrestrial sensitivity maps based on bedrock and surficial geology criteria. Detailed maps for eastern Canada south of 52 N and east of the Ontario-Manitoba border were generated by the Inland Waters and Lands Directorate (MOI, 1983). This mapping was revised for Nova Scotia (Hirvonen, 1984), Quebec (Li, 1985), and Ontario (Cowell and Lucas, 1986). Also available is a map for Labrador (Hirvonen, 1982). For comparison, aquatic sensitivity maps based on grouping the land by lake alkalinity are also available. One example is the "Acid precipitation and watershed sensitivity map" produced by the New England Secretariat and the Eastern Canada Secretariat (Committee on the Environment, 1988). This map includes all of eastern Canada (except Ontario), and several northeastern States (Connecticut, Maine, Massachusetts, New Hampshire, Rhode Island, Vermont).

TABLE 5.3.1

Potential to reduce acidity classes: soil and bedrock criteria [a]

Classes	Soil and Bedrock Criteria
High	All areas underlain by bedrock, such as limestone, dolomite, and other carbonate-rich rocks, which can readily reduce acidity.
	Shallow clay or carbonate-rich soils over bedrock having a moderate potential to reduce acidity.[b]
	Deep clay and/or carbonate-rich soils.
Moderate	Bedrock having a moderate potential to reduce acidity, exposed in at least 50% of area. This includes mafic and ultramafic rocks, sandstone with carbonate cement, and fine-grained sedimentary rocks such as shale.
	Clay or carbonate-rich soils over bedrock having low potential to reduce acidity and 50 to 75% exposure.
	Shallow, loamy and/or carbonate-poor soils over bedrock having a low or moderate potential to reduce acidity.
	Shallow carbonate-free soils over bedrock having a moderate potential to reduce acidity
	Shallow clay or carbonate-rich soils over bedrock having a low potential to reduce acidity.
Low	Bedrock having a low potential to reduce acidity exposed in 75% or more of area (e.g. rocks having a high silica content; such as granite and quartzose sandstone).
	Deep loamy and/or carbonate-poor soils.
	Deep or shallow sandy or loamy (carbonate-free or carbonate-poor) soils over bedrock having a low potential to reduce acidity and 50 to 74% exposure.
	Shallow sand or carbonate-free soils over bedrock having a low potential to reduce acidity.
	Deep sandy and/or carbonate-free soils.

[a] Lands Directorate, 1988.

[b] "shallow": less than 1 m.

5.3.2.2.2 Western and Northern Canada

All areas in western and northern Canada were mapped for aquatic and soil sensitivity (Western Canada-LRTAP, 1982, 1983, 1987). A series of maps resulting from both activities are available for each province and territory (Manitoba: Wotton and Haluschak, 1986; Saskatchewan: Padbury, 1986; Alberta: Holowaychuk and Fessenden, 1987; British Columbia: Weins, 1986 Northwest Territories: Government of Northwest Territories, 1986; Yukon Territory: Senyk, 1986). A further mapping effort involves the integration of soils, geology and aquatic systems in terms of combined "landscape sensitivity classes." Four classes are established. A summary is shown in **Table 5.3.2**.

TABLE 5.3.2

Western Canada: Percent distribution of landscape sensitivity classes, by province and territory [a]

Geographic Area	Sensitivity Class[b]				Total Land Area X10^6
	1	2	3	4	
(%)			(ha)		
British Columbia	9.2	1.7	26.0	59.6	93.1
Alberta[c]	1.1	1.0	10.7	87.1	66.1
Saskatchewan	14.0	0.6	22.3	63.0	65.3
Manitoba	2.7	0.1	31.9	65.3	64.9
Northwest Territories	22.7	2.9	24.4	41.5	297.0

[a] Sandhu and Lechner, 1988.

[b] Class 1: lake alkalinity < 10 mg/L, P low;
Class 2: lake alkalinity < 10 mg/L, P moderate;
Class 3: lake alkalinity > 10 mg/L, P low;
Class 4: lake alkalinity > 10 mg/L, P high.
P = potential of soil and geologic materials to reduce acidity.

[c] Tentative assignment, especially for the northeast portion of the province.

5.3.2.3 NATIONAL PERSPECTIVE

The results of sensitivity mapping based on the terrestrial rating scheme shown in **Table 5.3.1** are summarized in **Table 5.3.3**, by province and territory (Lands Directorate, 1987, 1988). About 4,000,000 km (46% of the country) are rated as highly sensitive. This includes most of Newfoundland and Quebec, much of Nova Scotia, north-central and northwestern Ontario, northern Manitoba and Saskatchewan, western British Columbia, southwestern and northern Yukon Territories, the eastern portion of the District of MacKenzie, the District of Keewatin and much of Baffin Island in the District of Franklin. Bedrock in many of these regions tends to be granitic, and the landscapes are dominated by frequent outcrops and shallow, coarse-textured soils.

Moderately sensitive areas are located within the Gaspé peninsula, New Brunswick, western Ontario, British Columbia, the Yukon Territories and the western part of the Northwest Territories. Not sensitive to acidic precipitation are the St. Lawrence Lowlands, the central and southern portions of Ontario and Quebec, the Prairies and several islands of the Arctic Archipelago.

Organic soils have not yet been rated. These soils, although acidic, tend to be well bufferred. The vegetation above, however, is sensitive to acidic deposition.

TABLE 5.3.3

Percent distribution of soil sensitivity classes (terrestrial projections) across Canada

			Sensitivity Classes [a]		
Province/Territory	**High**	**Moderate**	**Low** (%)	**Unrated (ice)**	**Unrated (organic)**
British Columbia	32	44	18	3	3
Alberta	6	11	70	<1	13
Saskatchewan	37	3	56	0	4
Manitoba	30	2	38	0	30
Ontario	34	20	20	0	26
Quebec	82	8	7	0	3
New Brunswick	31	49	12	0	8
Nova Scotia	54	33	13	0	<1
Prince Edward Is.	46	54	<1	0	<1
Newfoundland	56	30	4	0	10
Northwest Territories	48	25	18	6	3
Yukon Territories	43	35	17	3	2
Canada	46	21	23	2	8

[a] High potential of the land to reduce acidity = low potential sensitivity of surface waters to acidification (Lands Directorate, 1988).

5.3.2.4 FUTURE DIRECTIONS

First approximation maps describing the potential of soils and geology to reduce acidity are now available across the country. Current and future mapping efforts will likely enhance the resolution of the maps, and enable the application of different interpretations to existing base maps. Enhanced resolution will address the occurrence and distribution of aluminum, heavy metals and sulphates in surface waters, as affected by soils, surficial deposits, bedrock, and acidic deposition.

Re-interpretations will involve qualitative and quantitative ratings of soil sensitivity in terms of soil acidification, pH reduction, mobilization of exchangeable bases, and solubilization of aluminum in response to acidic deposition. Such interpretations should be useful for analyzing regional impacts on forestry and agriculture. Of further use will be the development and mapping of "critical loading" standards intended for the protection of ecosystems with different sensitivities to acidic deposition.

5.3.3 REMOTE SENSING

5.3.3.1 REMOTE SENSING APPLICATIONS OUTSIDE CANADA

Remote sensing has the potential to provide an objective, quantitative and standardized forest decline assessment methodology. Experts convened specifically to discuss the state of the art in remote sensing of forest decline met at a workshop in Austria in March 1988 (Nilsson and Duinker, 1988). This workshop concluded that the current technical problems were too great and the landscape, tree species, forest types and decline characteristics too varied to accomplish a reliable Europe-wide forest decline survey with existing remote sensing systems. However, significant success has been achieved using remote sensing to assess forest decline associated with some forest types in specific regions. For example, decline of the high-elevation spruce-fir forest on Vermont's Green Mountains in the northeastern U.S. has been accurately mapped using the Landsat Thematic Mapper (TM) and the airborne multispectral scanner Thematic Mapper Simulator (TMS). Multi-temporal images mapped changes in the forest's condition with time (Van Voris and Wukelic, 1988), whereas TM and TMS band ratios, specifically band 6/5 and 6/4, consistently and accurately delineated areas of spruce-fir decline (Rock et al., 1986; Vogelmann, 1988). Landsat TM was also used successfully in Sweden (Ekstrand, 1989) and North Carolina (Khorram et al., 1989) to map areas of light to moderate spruce decline. In both of these studies, areas of severe spruce decline could not be consistently detected. The authors concluded that the reason for the inconsistent classification was the reflection of ground vegetation through the severely defoliated spruce canopy, which masked the spectral reflectance of the spruce foliage. This problem was also experienced in a remote sensing project conducted in the Sierra Nevada range of California in the mixed conifer/yellow pine region (NCASI, 1989).

This project was designed to assess the spectral response of pondersosa pine needles to ozone as an early forest effects detection method. Although the results were promising, showing a good correlation between spectral response and chlorophyll content of needles, the need to develop approaches to address the mixing of signals from multiple environmental sources (rocks, soil, brush) were described. It was concluded that because of their high spectral resolution, the Advanced Visible Spectrometer (AVIRIS) and NASA's planned High Resolution Imaging Spectrometer (HIRIS) may provide better tools to detect early signs of forest damage.

Aerial photography has been used to assist forest decline surveys in Europe. These surveys often use medium scale (eg. 1:8000) colour infrared photography with a systematic network of photo sample plots in which single-tree assessments of decline symptoms are made. Supported by ground work, overall decline condition and distribution is determined on a regional basis. Experimental work has examined airborne multispectral scanner data at various resolutions. Classification of broad damage levels on high resolution systems has met with some success (Koch et al., 1984). In addition, spectral reflectance studies have concentrated on refining the spectral signature of declining trees in order to develop multispectral survey methods.

5.3.3.2 REMOTE SENSING APPLICATIONS WITHIN CANADA

In Canada, satellite remote sensing is routinely used for forest burn and cut-over mapping by Forestry Canada. Lee (1989) gives a summary of forest-related remote sensing research, which includes insect and disease damage, tree species and forest-type delineation and regeneration mapping. Low resolution (3 to 10 m) airborne multispectral scanner data have been moderately successful for insect damage assessment (Leckie and Gougeon, 1981; Ahern et al., 1986; Leckie and Ostaff, 1988). High resolution (<3 m) data have been used for forest stands affected by the mountain pine beetle and the spruce budworm.

Currently, forest decline in Canada is restricted almost exclusively to the eastern hardwood forests. Decline of coniferous forests has not, to date, occurred on a regional basis, although localized damage has been documented in the vicinity of point sources of air pollution. For example, remote sensing has been used very successfully to detect subtle effects of SO_2 on western hemlock around a sulphite pulp mill in British Columbia (Curtis and Ustin, 1989). Using a high resolution spectral sensor to generate spectral signatures along the SO_2 gradient, the authors found that a decrease in foliar chlorophyll content was related to a decrease in the depth of the chlorophyll absorption well (600-700 nm) and that chloroplast damage resulted in a broadening of the chlorophyll absorption feature. Large scale aerial photography has also been used effectively for assessing and quantifying SO_2 damage (Murtha, pers. comm.).

Colour infrared and standard air photography has been used to augment ground-based assessment surveys of sugar maple decline in eastern Canada. A test of Landsat Thematic Mapper imagery was conducted in Quebec to assess maple decline subsequent to the current decline episode south of the St. Lawrence River. However, only areas of severe crown defoliation could be detected (Beaubien, pers. comm.). Multispectral Electro-optical Imaging Scanner (MEIS) data were obtained in the same region of Quebec in the summer of 1989 and will be used to assess the effectiveness of this system for monitoring maple decline. The MEIS has been suggested as a good candidate sensor for forest decline assessment. However, inability to image in the short wave infrared and restricted swath width may reduce its potential effectiveness as an operational tool.

The Fluorescence Line Imager, a programable high resolution imaging spectrometer, has been used experimentally to compare spectral response between healthy and declining sugar maple sites in Ontario (Forestry Canada, 1988). The site with more extensive and severe sugar maple decline exhibited a significant shift towards shorter wavelengths at the red edge, relative to the site with healthier trees. This blue shift parameter is generally accepted as an indication of vegetation stress. However, two other theoretical spectral indicators of vegetation stress, an increase in the reflectance of the chlorophyll absorption well and a decrease in the reflection of the infrared shoulder height, were not observed in this preliminary study.

Sugar maple decline in Ontario has also been experimentally monitored with large-scale 4-band aerial multispectral videography (AMV) (Yuan et al., 1989). AMV was obtained over woodlots experiencing a gradient of sugar maple decline, in which tree condition assessment had been conducted by ground survey. Spectral and textural transformations were performed on the imagery and the resultant data were used to develop a numerical decline index on an individual tree basis, to parallel the ground-based assessment. The woodlots were then ranked according to their AMV decline index in relative order of sugar maple decline severity. The AMV decline index was consistent, objective, quantitative and compared favourably with the ground-based assessment. This technology is considered semi-operational.

There are several technical issues to be resolved before remote sensing of forest decline in Canada (and elsewhere) is operational. With low resolution imagery, confusion in damage assessment occurs as a result of mixed pixels (i.e., pixels containing trees of several species, open areas, or varying damage levels). With high resolution data, damage assessment confusion results from differing pixel intensities on an individual tree due to bidirectional reflectance effects (e.g., sunlight versus shade side of the tree).

5.3.4 EARLY DIAGNOSIS

5.3.4.1 BACKGROUND AND OBJECTIVES:

Forest decline is a major concern in parts of Europe including West Germany. Symptoms observed on affected trees are not reversible in many instances and this

decreases the potential for forest managers to react to the development of decline. In order to permit forest managers to take action before obvious symptoms of decline appear, it is necessary to have an appropriate means of detecting subtle changes in the characteristics of the trees. This led to the development of an "Early Diagnosis" project in which potential test methods could be evaluated. In 1986, a study was initiated in Europe to develop methodologies for measuring biochemical or physiological injury prior to the onset of visible injury symptoms or reduced rates of growth in forest trees. Numerous samples of tree foliage were collected across a pollution gradient in Europe and the samples were subjected to a variety of physiological and biochemical tests. The results of much of this work have been published (Cape et al., 1988; Mehlhorn et al., 1988; Wolfenden et al., 1988). These regional studies have been built on existing knowledge of how forests respond to pollutants emitted from point sources. One notable example has been the four-year (ongoing) intensive case study of Alberta boreal forest response to sulphur gas emissions including pollutant deposition and uptake, woody plant water status, physiology, mineral nutrition and biochemistry. The series of papers emanating from that study are summarized in Legge et al. (1981).

5.3.4.2 STUDIES IN ONTARIO

Some tests conducted in Europe in 1986 appeared to be useful in determining anatomical changes and alterations in foliage surface properties across an air pollution gradient. During 1987, some of the more promising methodologies employed in Europe were applied in Ontario to determine if there were any similarities between the situation in Europe compared to Ontario.

Samples of foliage from white pine, Norway spruce (a common species with Europe) and sugar maple were collected at 12 or 16 locations representing at least four distinct zones of deposition of acidic precipitation and ambient ozone across southern Ontario . The conifer foliage was divided into separate years of growth.

Measurement of the contact angle between water droplets and the surface of the conifer foliage was made under a high-power binocular microscope fitted with a protractor reticule. This test measures foliar wettability, provides an indication of cuticle integrity and has been used successfully to evaluate the air pollutant effects on leaf surfaces in both crops (Percy and Baker, 1988) and forest trees (Cape, 1983; Percy, 1987). The results indicated that changes to the cuticle were greatest (greatest degree of foliage wetting) in southwestern Ontario, the zone of greatest pollution deposition. Although conifer cuticles can respond directly to near ambient levels of acidity and ozone by producing new crystalline wax structures (Percy et al., 1989), gross cuticular changes by themselves cannot be considered reliable indicators of pollution deposition (Percy, 1988).

The Härtel (Fuchshofer and Härtel, 1985) test which indicates internal foliar changes following exposures to sulphur compounds was also examined. Absorbance of the foliar extracts indicated that turbidity of the solution increased with age of foliage and was greater in the zones of highest acidic deposition for both conifer species.

Changes in internal characteristics of the foliage related to pollution deposition were suggested by these tests.

Foliage samples were also assayed for 12 inorganic nutrients. Statistical differences were encountered between collection sites for most elements; however, there was no clear relationship between pollutant deposition and elemental composition. Rather, the properties of the local soil appeared to have had a greater influence in determining foliar composition. Age of foliage also had a definite role in elemental composition. Several elements including calcium, chromium, iron, magnesium, zinc, and lead were accumulated over time, whereas nitrogen and phosphorus concentrations decreased from year to year.

The study in Ontario included field sampling of local vegetation with similar tests being applied to plant material exposed to simulated acidic precipitation and ozone under controlled environmental conditions. The analysis of samples included pigments, chlorophylls A & B, carotene, lutein, and the xanthophylls, violoxanthin and antheraxanthin. The ratios of the latter have been suggested as indicators of tree health status in Europe (Wolfenden et al., 1988).Although the tests are ongoing, results have failed to show a clear relationship between pollutant deposition and any of the pigment variables. Inherent variability of samples collected in the field may prove to be too great to be of practical value in assessing tree health by this method. However, taken together with other tests still to be considered, analysis of pigment composition may still prove to be a useful diagnostic indicator of pollution-related stress.

The composition of spring sap collected from sugar maple trees at several locations in Ontario also was examined. Within stand comparisons showed that generally healthy trees produced sap of different quality compared with declining trees in the same stand. Healthy trees had higher concentrations of disaccharide sucrose, whereas declining trees had higher concentrations of monosaccharide sugars such as arabinose and galactose. Several inorganic elements were present in higher concentrations in sap from declining trees. These included aluminium, manganese, iron, sodium and barium. By contrast, calcium, potassium and boron were lower in declining trees. Full interpretation of the results from this part of the Ontario study is still underway.

Recent research in Alberta has shown that the ratio of accumulated foliar sulphur to assimilated sulphur is a useful biomonitoring tool in forest ecosystems under sulphur air pollution stress and might have application in early warning environmental management tool (Legge et al., 1988).

5.3.5 BIOMONITORING

Biomonitoring studies have long been utilized in studying the indirect effects of local sources of air pollution, and some of this work has been adapted to LRTAP pollutants. In addition, these types of studies are frequently used as monitoring tools, as a surrogate for other types of air pollution monitoring, to determine the

relative uptake of LRTAP contaminants over different geographic areas (Zakshek et al., 1986; Tims and Knight, 1987).

Lichens and mosses are particularly good subjects for this type of research because of their simple tissue organization, their tendency to retain and accumulate ions, and their relative sensitivity to pollutants. Recently, higher plants and physiological indices have also been used as indicators of LRTAP effects.

Biomonitoring studies are of two basic types: bioindicator studies which use sensitive indices of biological health (visible or physiological responses) to assess exposure to levels of toxic or harmful contaminants and effects on the vigour of ecological communities; and bioassimilation studies which measure accumulation of contaminants in plant and animal tissues as an indication of pollutant exposure.

5.3.5.1 BIOINDICATOR STUDIES IN CANADA

Bioindicator studies in Canada have primarily used lichens, mosses or higher plants to monitor the effects of LRTAP pollutants on the environment. Although there have been laboratory studies designed to measure directly the acidic rain tolerance of lichens and mosses (Lechowicz, 1982), much of the recent Canadian work in this field has been done in natural systems. In the boreal forest of Ontario, researchers have studied the effects of simulated acidic rain on permanent field plots of Cladina lichens (Scott et al., 1989) and feather moss (Pleurozium schreberi) (Hutchinson and Scott, 1988). Lechowicz (1987a) has monitored the effects of simulated acidic precipitation on Cladina stellaris in subarctic lichen woodlands of northern Quebec.

Higher plants in boreal ecosystems have also been used as bioindicators. Redmann et al. (1986) used simulated acidic rain to determine the relative sensitivity to acidification of plant species of the southern boreal forest in Saskatchewan. Membrane leakage differed significantly among 16 species tested, and can be used to categorize resistance to acidification of leaf tissue.

A principal component of the Acid Rain National Early Warning System (ARNEWS) is the use of various biological indicators of plant health, especially tree vigour (Magasi, 1988). In British Columbia, lower plants have been used as an additional measure of change in ARNEWS plots; included are an inventory of lichens, liverworts and mosses and an assessment of the vigour of transplants of pollution sensitive lichens (Van Sickle, 1989). Additional lichen biomonitoring networks are now being established in B. C.

Garvais et al. (1989) have studied the chromosomal characteristics of Claytonia caroliniana to find an early indicator of deterioration in maple stands. The occurrence of B chromosomes has been shown to be negatively correlated with the severity of maple dieback in the stands studied.

The Ontario Ministry of the Environment also is developing methodologies for

measuring biochemical or physiological injury prior to the onset of visible injury symptoms. This work is described in greater detail in Section 5.3.4 - Early Diagnosis.

The New Brunswick Department of the Environment uses ozone sensitive Bel W-3 tobacco, co-located with conventional ozone monitors, in order to assess ozone concentrations and the relationship between ozone concentrations and white birch deterioration in the Bay of Fundy area (Tims and Knight, 1987; Cox et al., 1989).

5.3.5.2 BIOASSIMILATION STUDIES IN CANADA

Ontario Ministry of the Environment (Case, 1985) used several species of Cladina, and the mosses Hylocomium splendens and Pleurozium schreberi to map the deposition of trace elements at 44 locations across Ontario, and to investigate the relationship between lichen and moss chemistry and precipitation chemistry. Barkley (1989) has undertaken a study of trace metals in lichens and mosses in the Thunder Bay District. Zakshek et al. (1986) compared levels of S and Pb in the lichen Cladina rangiferina at 85 ombrotrophic peat bogs in eastern Canada. In the Maritime provinces, Percy (1983) determined patterns of atmospheric deposition for the elements Cd, Co, Cr, Cu, Fe, Hg, Mn, Ni, Pb, S, and Zn at 61 ombrotrophic bogs, using the moss Sphagnum magellanicum. In New Brunswick, trace metal deposition to species of Sphagnum and Cladina is being measured in bogs (Tims, 1989).

Although lichens and mosses have traditionally been used for this type of work, analysis of tree foliage is becoming more commonly used as an indicator of bioassimilation. The Ontario Ministry of the Environment has undertaken a baseline foliar chemistry survey in order to monitor changes and trends in tree foliar constituents (McIlveen, 1989). Components of the ARNEWS study also include measures of the chemical composition of lichens, soils and foliage to monitor long term changes (Magasi, 1988).

Several jurisdictions have monitoring programs to determine the content of trace metals in organs of herbivores. These are discussed in greater detail in Section 5.4.3

5.3.5.3 CONTRIBUTIONS TO THE KNOWLEDGE OF CONCENTRATIONS AND DEPOSITION OF LRTAP POLLUTANTS

Bioindicator studies have contributed considerably to the knowledge of the levels of LRTAP pollutant deposition and impact on terrestrial and other ecosystems.

As well as providing data on the probable concentrations of ozone in rural and forested areas, ozone biomonitoring results from New Brunswick, Quebec, Ontario and British Columbia should be of considerable assistance in the review of ozone

ambient air quality objectives and the possible establishment of a long-term, growing season plant response index for ozone exposure.

Studies of the trace metal content of lichens and mosses have greatly increased the knowledge of patterns of trace metal deposition to rural areas of Canada (Zakshek et al., 1986; Case, 1985; Percy, 1983). For sulphur deposition, both receptors showed highest values in central Ontario and Quebec, with distinct east-west gradients. Lead did not show a distinct east-west gradient; however, lowest concentrations were in Newfoundland and greatest concentrations were in southern Quebec and Ontario. Working in the smaller geographical area of the Maritimes, Percy (1983) showed no concentration gradients of a regional scale in the metal content of Sphagnum, but identified localized zones of enrichment, associated with local emissions.

One of the principal challenges of these studies has been to relate these measurements to direct deposition measurements by chemical and physical means. Zakshek et al. (1986) found that concentrations of S and Pb in the lichen Cladina rangiferina in eastern Canada compared well with levels measured in lakes sampled on an east-west axis and with precipitation chemistry results. Case (1985) has demonstrated that, in Ontario, Cd and Pb in lichens and bulk precipitation are significantly correlated, although a significant amount of the variation in trace metal content of the plants was not explained by precipitation chemistry.

The observation that some trace metal levels in arctic ecosystems may be elevated is also of concern (Puckett and Finegan, 1980) and it may be necessary to continue monitoring of metals in these systems to determine whether or not increases have occurred within the last decade.

Atmospheric sources contribute to the food chains of wildlife that have accumulated elevated levels of metals and other contaminants (Section 5.4.3) Metal mobilization due to acidic precipitation impacts also may be a factor; cadmium levels in deer and moose and their preferred forage plants tend to be higher in poorly buffered, acid sensitive areas in Ontario (Glooschencho et al., 1988; Parker, 1989; Hickie et al., 1989), and, in Quebec, were found to be lowest in zones where the soils indicated low sensitivity to acidic precipitation (Crete et al., 1987).

5.3.5.4 CONTRIBUTIONS OF BIOMONITORING STUDIES TO THE KNOWLEDGE OF LRTAP EFFECTS

Studies of the effects of simulated acidic precipitation on boreal forest understory species have shown that species responses are variable and complex (Scott et al., 1989).

Some species, including Cladina rangiferina, are tolerant of rainfall of less than pH 3.5; percent cover of Vaccinium angustifolium actually increases over the short term under simulated rainfalls of pH as low as 2.5. Cladina stellaris appears to be

resistant to additions of simulated acidic rain (Lechowicz, 1987a). In most cases, however, increased acidity of rainfall results in reductions in growth, size and/or cover of understory cryptogams, including some species of Cladina (Scott et al., 1989) and the feather moss, Pleurozium schreberi (Hutchinson and Scott, 1988).

This general trend towards reduced productivity and growth in boreal forest lichen and moss understory species exposed to repeated simulated acidic rain episodes suggests that occasional rainfall events of pH 3.5 or lower during the growing season pose a threat to the health of the species and to the microhabitat of the overstory and understory community structure (Scott et al., 1989). Observations that acidic rain has a direct effect on the growth and vigour of lichens has serious implications, since lichens play a substantial ecological role in boreal and especially arctic and subarctic ecosystems, comprising a major winter food source for ungulates, controlling water balance through their mulching effect and improving the nitrogen balance of subarctic soils (Lechowicz, 1982).

Studies of higher plants in boreal systems have shown that these plants also vary in their sensitivity to acidification. In studies at the University of Saskatchewan, broadleafed species were shown to be more sensitive than conifers (Redmann, 1989).

Studies on the bioassimilation of trace metals by herbivores suggest that trace metals are being accumulated in the food chain. As these levels approach toxic concentrations, animals at the top of the food chain are at risk. Results of these studies have prompted officials in Manitoba, Ontario, Quebec, New Brunswick and Newfoundland to post health advisory statements that people should not consume the organ meat of affected species (Wotton and McEachern, 1988; Glooschencho et al., 1988; Crete et al., 1987; Ecobichon et al., 1988; Brazil, 1989).

5.3.5.5 THE FUTURE DIRECTION OF BIOINDICATOR STUDIES

Around strong point sources, it has been possible to establish specific dose response relationships between pollutants of concern and the health responses of biological components; however, in the case of LRTAP, specific responses of forest ecosystems to pollutants have proven more difficult to document. Research into specific responses of both lower and higher plants will have to continue before these responses can be used with any confidence in assessing the inputs and impacts of LRTAP pollutants.

Despite this difficulty, there is an increasing need to establish biomonitoring systems that will detect change in forest, wetland, and grassland systems before these systems are irreparably damaged. This will result in continuation of work toward the development and sophistication of early warning systems, discussed elsewhere in the document.

5.4 EFFECTS

5.4.1 FORESTS

5.4.1.1 POTENTIAL DIRECT EFFECTS

5.4.1.1.1 Acidic Deposition

The potential effects, direct and indirect, of acidic deposition on vegetation were described in the late 1970's by Tamm and Cowling (1977). Four key areas of potential impact were described as follows:

a) Damage to protective surface structures.
b) Interference with guard cell function.
c) Disturbance of normal metabolism or growth processes
d) Accelerated leaching of substances from foliar organs.

Effects on the Protective Surface-Cuticle

The cuticle of the tree leaf/needle is its primary defense from biotic and abiotic stress, preventing water and nutrient loss, and invasion by pathogenic organisms. It is not surprising that a considerable effort has gone into understanding the relationship between pollutant deposition and the maintenance of cuticle integrity. While gaseous pollutants are free to interact with the cuticle and invade the leaf through stomatal pores, acidic wet deposition (droplet), must act primarily by its interaction with the cuticular surface. This can occur through the leaching of soluble substances or, in the case of severe acidicity, through the collapse of epidermal cells. Wettability of the cuticular surface, then, is considered to be important in determining the susceptibility of foliage to acidic deposition.

Exposure of plants to air pollution in the field has been shown to alter the epicuticular wax structure of conifers and thereby increase the wettability of the foliage (Cape, 1986; Percy and Riding, 1978). Simulated acidic rain (SAR) treatment of needles in the laboratory has resulted in the production of amorphous structures within the epistomatal chambers (Percy, 1987; Rinallo et al., 1986) and in the erosion of surface waxes (Baker and Hunt, 1986). Recent experimental evidence has indicated that acidic rain and fog applied at ambient levels to elongating red and Sitka spruce needles alter epicuticular wax structure by interacting directly with wax biosynthesis rather than through a physiological erosion or weathering (Percy and Baker, 1988; Percy et al., 1990). The implications of these structural changes to the functioning of the tree are not well understood at present. Suggestions have been made that these changes bring about the decreased capability of the tree to prevent invasion by pathogens, or to prevent leaching of substances from foliage; however, there is insufficient evidence, at the present, to confirm that this is the case.

Effects on Guard Cell Function - Photosynthetic Response

The effect of acidic wet deposition on guard cell functioning would seem to be a research area of considerable importance; however, it does not appear to have received much attention in the literature. Photosynthesis has been investigated by Reich et al. (1986) in sugar maple and red oak following treatment with pH 3.0. SAR. They could find no significant effect of the most acidic treatment (pH 3.0) in terms of a reduction in net photosynthesis. While the authors did not investigate stomatal response per se, one might conclude from their photosynthetic data that stomatal response, at the least, was not influenced to the point where it had any impact on net photosynthesis. Neufeld et al. (1985) investigated the effects of SAR on stomatal conductance and net photosynthesis in Liquidambar and Robinia. Significant treatment effects were found only in trees exposed to SAR pH 2.0.

Effects on Reproduction and Growth

In contrast to several other characters tested, it has been established that ambient wet deposition is often acidic enough to inhibit pollen germination in most forest plants tested (Cox, 1983). Hardwood pollens are generally more sensitive in this respect, with 50% reduction in germination of their pollen grains at pH's of 4.2-3.6, the average pH of rain in some regions. This level of inhibition in conifers is not reached until the pH is as low as 3.6-2.8, the range of only the more acidic rain events. These lower pH's are common or average values for fog and cloud water intercepted by vegetation in some areas (Schemenauer, 1986; Kimball et al., 1988; Cox et al., 1989). The conservative nature of the variation in pollen response in birch suggests the importance of pH in pollen stigma interactions in this genus. Environmental modification of stigmatic surfaces with either gaseous pollutants or acidic wet deposition will lead to disruption of pollen function and affect eventual seed set (Cox, 1988; Venne et al., 1989).

The interaction between acidic precipitation and tree growth has been documented for several species. Studies by Abrahamson et al. (1975) in Norway in which Lodgepole pine, Scots pine, and Norway spruce were exposed to SAR at pH 3.0, or 2.0, failed to show a reduction of growth in terms of height or diameter increment over a four year treatment period (Tviete and Abrahamsen, 1980). Similarly, studies with red oak and sugar maple over one growing season failed to show a relationship between several growth related variables (height increment, stem diameter, total dry weight) and the acidicity of rainfall,down to pH 3.0 (Reich et al., 1986). Growth reductions in yellow poplar (Dochinger and Jensen, 1985) were observed following treatment with SAR in terms of height, leaf dry weight, and leaf area. These reductions only occurred, however, when the SAR acidity was pH 2.5. Reich (1987) examined the effect of SAR down to pH 3.0 on some growth characteristics in white

pine. He found that there were no reductions in growth associated with increasing acidity and that needle and stem growth were stimulated (47 and 35 %, respectively) by increases in acidity to pH 3.0 on three soils tested. Similarly, a study of the effects of SAR on four hardwoods by Neufeld et al. (1985) found no significant effect of acidity down to pH 3.0 on root, stem, leaf or total biomass. Taken together, these data suggest that direct growth effects are recorded only when the pH of precipitation falls below a critical pH value, possibly pH 3.0. Above this value, the fertilizing effect of additions of N, in particular, and S through the acidic treatment appear to override any effect of hydrogen ion addition, at least in the short term.

Effects on Nutrient Loss

Early studies on the effects of precipitation on cation leaching from foliage (Tukey and Mecklenburg, 1969; Tukey, 1970) suggested that loss of cations from foliage occurred with considerable ease. A logical extension of this reasoning was that increased acidity would result in accelerated cation loss. While it is generally true that cation loss does increase in relation to increased acidity (Wood and Bormann, 1975), the magnitude of the response may be influenced by species, nutritional status and environmental conditions. In addition, not all cations are leached to the same extent. Increased loss of calcium and magnesium has been reported in response to an increase in external acidicity at pH 3.0 in Nicotiana (Fairfax and Lepp, 1975) and sugar maple (Wood and Bormann, 1974; Hogan and Foster, 1989). Losses at pH's closer to ambient, pH 4.5 for example, would be much less. It is readily apparent from these studies that potassium, an element of considerable interest in relation to tree decline, was not as readily leached at pH's close to ambient as was calcium. The interim conclusion is that foliar cation leaching is only likely to be of significance where available soil supplies of these cations approach limiting concentrations.

Summary

Most studies have examined the effects of simulated acidic wet deposition as a single pollutant in terms of its effects on isolated morphological or physiological indicators. While there are exceptions, generally these indicators do not respond in a negative sense until the pH is close to 3.0. The threshold for visible foliar injury in most species examined has been shown to be close to this value or at some value that is more acidic. The apparent lack of injury at pH's close to current ambient was demonstrated by Jacobson (1984). He showed that there was little overlap between the pH of ambient rainfall events at Ithaca, N.Y. and the reported pH for foliar injury to vegetation. This suggests that the possibility of direct damage to vegetation by ambient rainfall is slight. However, evidence is not yet available to assess the possibility of interaction between acidic wet deposition and gaseous pollutants or biotic stresses at the foliar level.

5.4.1.1.2 Gaseous Pollutants

Several phytotoxic gases such as O_3, SO_2, NO_x, PAN and possibly H_2O_2 and VOC are known to be either precursors of or associated with acidic wet deposition. Even though acute (symptomatic) negative effects on forest vegetation have been shown for each of these gases under controlled conditions at elevated concentrations, their effects at ambient levels are still a matter of debate. Most of the problem lies in the confusion made in the literature involving the use of notions such as "stress" and "damage". Based on this consideration, Keller (1977) has developed the concept of "latent injury" caused by pollutants. Latent injury usually results from sub-lethal doses of pollutants which while impairing some cell functions or chemical balances within a leaf or needle, is not manifest as visible lesions or other foliar discoloration. This concept is important as such injury relates to the current ambient air quality in forested areas and may be critical in the predisposition of foliage to other abiotic or biotic stresses.

Relative Importance of Gaseous Pollutants

According to McLaughlin (1985), direct effects of pollutants on vegetation, in the U.S., would be expected to occur across large areas mainly from O_3. In contrast, effects of SO_2 would be anticipated in industrial areas close to point sources. However, little is known of the effects at ambient levels of other secondary gaseous pollutants such as VOC, H_2O_2 and PAN.

In North America, annual average levels showed that O_3 occurs at phytotoxic concentrations, usually one order of magnitude larger than SO_2 and NO_x (NAPAP, 1988). For that reason and also based on increasing concentrations in the troposphere,(about a doubling in Europe since 1956) (McLaughlin, 1985), O_3 is becoming a major concern on a world-wide scale (Krupa and Manning, 1988).

Physiological Impacts

The ability of ozone to induce acute injury has been well documented, with numerous studies with trees attesting to its ability to cause growth reductions. Its effect at near ambient levels experienced in eastern Canada, however, has only received attention within the last few years. As with the other gaseous pollutants, O_3 penetrates the plant mainly through the stomata. Once inside the leaf, it is free to interact with many systems, but its major effect appears to be in the reduction of photosynthesis. Reich et al. (1986) demonstrated reductions in net photosynthesis in several tree species at ambient or near ambient concentrations of O_3. These reductions were shown to be related linearly to concentration and dose. At no time did the foliage show "classic" symptoms of O_3 injury.

Generally, factors affecting stomatal conductance, such as relative humidity, water availability and leaf developmental stages also influence the plant's response to pollutants. Reich (1987) has proposed a model suggesting that plants with high stomatal conductance, like agricultural crops, would be more susceptible to pollutants than trees. Even though the response could vary from one species to another, Reich (1987) showed that net photosynthesis tends to decrease linearly with the O_3 dose that a plant is absorbing (i.e., effective dose).

However, exposure to chronic doses of air pollutants does not generally cause linear reductions in plant growth (Mooney and Winner, 1988). Even if the conceptual link between a reduction in dry matter accumulation and a lower carbon gain via reduced photosynthesis seems to be obvious, other physiological activities such as carbohydrate translocation and partitioning, also respiration, must be considered to accurately assess how pollutants affect plant growth (Mooney and Winner, 1988). It has been shown that carbon metabolism and translocation are influenced by O_3 and H_2O_2 (Koziol et al., 1988). Therefore, air pollutants could create allocation shifts within plants. Even though only a limited number of studies have yet been conducted, it seems that O_3 has the general effect of reducing carbon allocation to roots (Mooney and Winner, 1988; Lechowicz, 1987b).

Direct effects of O_3 on cuticle permeability have also been previously hypothesised (Prinz et al., 1984). It was proposed that O_3 could stimulate mineral leaching from the leaves via increased cuticular permeability. Garrec and Kerfourn (1989) have shown however, that the action of O_3 does not interfere with cuticle permeability directly. Rather, they have suggested that it could modify the plant metabolism related to cuticle formation.

Reduced cone size, seed weight and quality, together with lowered pollen viability have been observed adjacent to industrial and urban areas both here and in Europe (Houston and Dochinger, 1977). Natural variation between trees in pollen sensitivity to acidity and gaseous pollutants (SO_2 and 0_3) have been determined (Cox, 1989; Venne et al., 1989). Given the degree of genetic overlap apparent (60%) between gametophyte and sporophyte in plants, selection in the sporophyte (pollen) population for pollution tolerance may be adaptive if gametophyte and sporophyte environments correlate. If however, no such correlation exists, reduced fitness may result. In addition, this variation in pollen response may lead to changes in fertility selection (male fitness) in forest tree populations, affecting genetic composition. This, together with the direct selection for pollution tolerant genotypes in tree populations and potential losses in germplasm, may be a more important impact than short term economic losses (Karnosky et al., 1989).

Summary and Implications

Most of the studies to date involve seedlings exposed to pollutants in modified environments (i.e., open-top chambers, growth cabinets, etc.). Thus even if evidence indicates that trees could be adversely affected without visible effects by

chronic O_3 exposure, the transposition of these results to mature trees must be treated with caution. Furthermore, potential interactions between different pollutants or natural stresses also increase the complexity of the evaluation.

Immediate environmental concerns demand that we understand the nature of tree responses to air pollution, including the possibility that current declines are influenced, in some way, by air pollution. Yet it is clear that our knowledge of the effects of near ambient concentrations of air pollutants, in combination with other stresses, is incomplete. The indirect impact of these combined stresses on the defense mechanisms of trees to other biotic or abiotic stresses is an important and growing field of research. Although much greater attention has been given to these interactive stresses for crop species, recent advances in the area of pollutant:insect interactions on trees are summarized by McNeill and Whittaker (1989) and Hain (1987).

Integrated studies involving fewer, but more comprehensive, approaches have been proposed by Koziol et al. (1988). It is hoped that a more comprehensive analysis would provide invaluable information and increase our understanding of plant adaptation to and recovery from anthropogenic environmental stress (Koziol, et al., 1988).

5.4.1.2 POTENTIAL INDIRECT EFFECTS

In the section on potential indirect effects, five topics will be discussed in detail:

- Soil acidification and cation leaching
- Aluminum toxicity
- Aluminum interference with calcium and magnesium uptake
- Excess nitrogen deposition and nutrient imbalances
- Terrestrial and aquatic linkages

5.4.1.2.1 Soil Acidification and Cation Leaching

In the few forest soils where information on changes in soil pH over time is available, decreases in soil pH have been observed over several decades. Exceptions, where soil pH remained unchanged over time have also been reported (Nowack et al., in press). Soil changes have been noted in pristine areas as well as those affected by acidic deposition (Johnson and Taylor, in press). Plant uptake of base cations also contributes to soil changes and the removal of phytomass (wood, needles and especially litter for animal bedding) from forest ecosystems, a long-standing former practice in many areas of central Europe, has also contributed significantly to the reduction in the supply of exchangeable bases in soil and has contributed to soil acidification (Ulrich, 1987).

Nutrient budget analyses suggest that atmospheric strong mineral acids are contributing to the accelerated leaching of base cations from some Canadian soils (Foster, 1985). Despite considerably greater acidic deposition to high elevation boreal forests in the eastern United States, as a result of cloud and fog acidity, soils at high elevation sites exhibit generally similar levels of acidity and exchangeable bases to comparable soils from low elevation forests and boreal forest soils in low deposition areas (**Table 5.4.1**).

High elevation soils may have reached a stage where base cation losses from the soil are greatly reduced because of the great difficulty in displacing them from exchange sites that are inherently low in these bases. It is also possible that the current rates of mineral weathering in these relatively young, glaciated soils may be sufficient to replace those bases removed from the soil; however, this remains unproven. Better estimates of the rate of weathering of primary minerals within the rooting zone of trees is critical to evaluating the consequences of cation leaching from soils.

One of the longest records of atmospheric deposition and element flux through a terrestrial ecosystem in Canada is the dataset from the Turkey Lakes Watershed in Ontario. Sulphate deposition, flux through the hardwood forest and export from the watershed via forest streams has been monitored for 10 years (1980-1989). A single-year decline in SO_4^{2-} deposition below the canopy from 34 kg/ha/yr in each of the 1982 to 1984 hydrologic years to 24 kg/ha/yr in 1985 was reflected in solutions collected at different depths in the soil. Controlling emissions to reduce SO_4^{2-} inputs to forested watersheds therefore is likely to reduce SO_4^{2-} leaching from podzolic soils similar to those at Turkey Lakes.

5.4.1.2.2 Aluminum Toxicity

Soil solution aluminum is controlled by the concentration of acidics and acidic anions applied to the soil and the ability of the soil to neutralize acidics. Aluminum in solutions in strongly acidic soils will be low because of the mitigating effects of organics. Aluminum is only likely to increase in mineral horizons in the presence of mineral acidic anions (Johnson and Taylor, in press). In Europe, excessive strong mineral acidic anion leaching is considered to have contributed to soil acidicification and the mobilization of Al^{3+}, which can contribute to a reduction in plant uptake of calcium and magnesium (Huttermann and Ulrich, 1984). Increased availability of aluminum in soil solution relative to divalent cations can be effected by either soil acidicification or simply by increasing the ionic strength of the soil solution by nitrate and sulphate deposition.

Table 5.4.1
Soil Properties[1] from Eastern North American Coniferous Forests[2]

Location: Lat. (N)	Location: Long. (W)	Elevation	Vegetation	pH (H_2O)	C	C/N	CEC[3]	Exchangeable Cations: K	Exchangeable Cations: Ca	Exchangeable Cations: Mg	Na	Base Saturation	Solution SO_4^{2-}
		m			g kg^{-1}		c mol, kg^{-1}					%	u mol L^{-1}
					Low Deposition								
50°02'	92°32'	370	Jack pine	4.9	30	35	7	0.36	0.71	0.17	-	18	-
48°21'	89°29'	460	White pine	5.4	14	23	(3)	0.24	1.91	0.28	-	(88)	85
48°31'	89°29'	460	Balsam fir	5.6	28	33	(5)	0.31	3.40	0.82	-	(93)	70
49°30'	87°50'	400	Black spruce	3.9	125	28	27	0.20	0.37	0.41	0.04	4	-
					Moderate Deposition								
45°07'	78°50'	-	White pine hemlock	4.5	60	27	(3)	0.17	0.79	0.13	-	(40)	120
47°19'	71°09'	640	Balsam fir	4.4	60	17	33 (15)	0.17	0.72	0.24	-	3	-
43°59'	74°14	530	Spruce-fir	3.6	140	-	37	0.08	0.41	0.22	0.02	2	120
					High Deposition								
440°	74°	1000-1500	Spruce-fir	4.1	85	26	(10)	0.10	0.40	0.10	0.20	(6)	170
35°45'	82°15'	1700'	Spruce-fir	4.3	200	26	36	0.29	0.70	0.38	0.28	2	65

[1] Average values from surface horizons to 30 cm depth; organic layers included if available
[2] Cited from Anon., 1987; Foster and Morrison, 1987; Friedland, (pers. comm.); Joslin et al., 1987; 1989; Lazerte (pers. comm.); Lozano et al., 1987; Mollitor and Raynal,1982; Morrison et al., 1976; Robitaille and Boutin, (pers. comm.).
[3] CEC by buffered NH_4OAC or in brackets unbuffered chloride salts (effective base saturation)

Table 5.4.2
Soil Properties[1] from Eastern North American Mixedwood and Hardwood Forests[2]

Location: Lat. (N)	Location: Long. (W)	Elevation	Vegetation	pH (H_2O)	C	C/N	CEC[3]	Exchangeable Cations: K	Exchangeable Cations: Ca	Exchangeable Cations: Mg	Na	Base Saturation	Solution SO_4^{2-}
		m			g kg^{-1}		c mol, kg^{-1}					%	u mol L^{-1}
					Moderate Deposition								
47°03'	84°24'	350	Sugar maple	4.5	45	16	15 (7)	0.12	1.20	0.25	0.22	12 (23)	55
46°	74°	380	Tolerant hdwd	5.1	71	-	(4)	0.12	1.66	0.20	0.05	(53)	60-70
45°25'	79°10'	-	Tolerant hdwd	4.4	48	19	(4)	0.18	2.40	0.43	-	(78)	80
44°22'	65°12'	130	Mixedwood	4.4	30	27	9	0.16	0.21	0.17	0.10	7	-
43°59'	74°14'	530	Tolerant hdwd	4.1	95	-	25	0.13	1.14	0.19	0.01	6 (25)	105
44°	74°	700	Tolerant hdwd	3.9	-	-	11	0.06	0.66	0.10	0.01	7	105

[1] Average values from surface horizons to 30 cm. depth; organic layers included (if available)
[2] Cited from Cronan, 1985; Foster et al., 1986; Hendershot, (pers. comm.); Lazerte, (pers. comm.); Lozano et al., 1987; Mollitor and Raynal, 1982; Percy, (in press).
[3] CEC by buffered NH_4OAC or in bracketed unbuffered chloride salts (effective base saturation).

The highest soil solution concentrations of soluble aluminum are characteristic of sites with soil less than 10 to 15% effective base saturation, with a pH < 4.9 and soil solutions with sulphate concentrations > 80 u mol L^{-1} (Cronan, in press). Reuss (1983) showed that minor changes in base saturation can produce large increases in Al^{3+} concentrations in soil solution for soils of 10 to 20% base saturation. Some of the soils of the Canadian shield with coniferous vegetation appear to be at risk because they contain pH and base saturation levels near or below the suggested critical levels **(Table 5.4.1)**. Soils under hardwood forests are more likely to leach bases when exposed to acidic deposition because their base saturation levels exceed critical levels **(Table 5.4.2)**.

In the strongly acidic soils compared here, the cation exchange capacity (CEC) is measured in a solution buffered at pH 7.0. The exchange capacity of organic matter is pH dependent and organic matter represents a major portion of the CEC of many of these forest soils that are low in clay. The CEC measured at or near field pH are only 1/4 to 1/2 those determined by the buffered neutral salt solution (NH_4OAc) (**Table 5.4.1, 5.4.2**; see also Kalisz and Stone, 1980). It follows that the effective base saturation of these soils would be two to four times greater (see examples in brackets). Projected effective base saturation levels in these soils still suggests that Canada has nutrient poor forest soils where either an accelerated loss of bases or aluminum mobilization is a concern.

Cronan (in press) has reported that rooting of red spruce and sugar maple was moderately sensitive to aluminum (e.g., from < 200 to 800 u mol L^{-1}) when seedlings were tested under experimental conditions. American beech was insensitive and red oak varied in sensitivity for different seedling groups. They examined species and levels of Al in soil solution collected by lysimeters across a selected number of sites in Europe and North America and found that the highest soluble Al concentrations were found in northern spodisols (15 to 70 u mol L^{-1}) and West German inceptisols (80 to 240 u mol L^{-1}). They also discuss the considerable uncertainties that exist in relation to defining thresholds for Al toxicity under field conditions. Not the least of these is the fact that lysimeters sample free draining water (< 10 kPa), whereas higher ion concentrations may be observed in plant available water (10-5000 kPa). Further, peak concentrations of sulphate, acidicity and Al in solution are likely to exceed the mean growing season (June to October) values presented in **Table 5.4.2**.

Although soil problems related to increased availability of Al have not been identified in all areas where tree decline is experienced in Canada, they have been implicated in high elevation declines in the northeastern United States (Shortle and Smith, 1988) and they have been associated with some, but not all European declines. In the Muskoka area of Ontario (McLaughlin et al., 1989), soil and sugar maple reet chemistry results also suggest a relationship with tree decline.

5.4.1.2.3 Aluminum Interference with Calcium and Magnesium Uptake

It has been hypothesized that the long-term consequences of leaching of base cations will be the development of nutrient deficiencies in soils (Abrahamsen, 1984), that may lead to nutrient imbalances in forest vegetation. Such losses may contribute to nutrient deficiencies on sites where the soil nutrient status is low (Joslin et al., 1989). For example, magnesium deficiency in Norway spruce foliage has been identified on acidic soils at high elevations in southern Germany (Zottl and Huttl, 1986). Fertilization has proven successful in reversing these foliar symptoms. Bernier et al. (in press) have reported K, P and Mg stress symptoms in declining sugar maple stands in southeastern Quebec and suggest that nutrient deficiency may be an inciting factor in damage occurrence in some areas. Since there is some evidence that the development of foliar nutrient deficiencies in sugar maple, cannot be predicted by examining soil properties alone (Hendershot, 1989), other factors must contribute to symptom development. One possibility is that high soil solution concentrations of Al, induced by acidic wet deposition, prevent uptake and utilization of base cations by the plant.

Experimental studies summarized by Cronan (in press) have revealed that "nutritional effects from Al exposure generally occurred at lower soluble concentrations (100 to 300 μ mol L^{-1}) than those associated with growth reductions." In the southern Appalachians, soils of high elevation spruce-fir forests are acidic (effective base saturation 5 to 10%), and soil solution Al concentrations approach threshold values for inhibiting uptake of Ca, Mg and P by red spruce (Joslin et al., 1987). There is a lack of agreement on whether this theory is a plausible explanation for spruce decline at high elevations. Circumstantial links between red spruce growth, changing plant chemistry and air pollution have been suggested by (Shortle and Smith, 1988; Bondietti et al., 1989 and McLaughlin et al. (in press)). Shortle and Smith (1988) interpreted the suppression of cambial growth and low Ca/Al ratios in fine roots of mountain-top spruce in Vermont as evidence of Al-induced Ca deficiency. Low Ca/Al ratios in high elevations, southern Appalachian spruce/ fir foliage, likewise suggest that co-deficiency stress is present in the trees (Robarge et al., 1989). Johnson and Siccama (1983), on the other hand, reported that spruce mortality and Ca/Al ratios in fine roots increased and Al concentrations in foliage and fine roots decreased with increasing elevation.

In general, linkages between pollutants and nutrient deficiency have not been proven. Extrapolation of the hypothesis on nutritional effects from Al exposure to species other than red spruce is unwarranted. Any relationship between soil-mediated effects and altered chemical and physiological response by the trees is inferred and the mechanisms that may produce possible damage are largely poorly understood.

5.4.1.2.4 Excess Nitrogen Deposition and Nutrient Imbalances

Although light deposition of N (5 to 15 kg ha^{-1} yr^{-1}) is generally regarded as beneficial to forests, higher loadings may be detrimental. Excess N deposition may produce, in the long-term, nutrient imbalance in some forests, such as Mg deficiency in Mg-poor soils (Zottl et al., in press) that may result in growth limitations. Excess inorganic-N availability in soil could reduce fine root biomass and produce a loss of ectomycorrhizal symbionts, thereby decreasing the uptake of water and P by the trees (Aber et al., 1989). In the southeastern Netherlands, annual atmospheric deposition of 50 to 100 kg õN ha^{-1} has produced excess ammonium availability in soil, to the extent that Scots pine trees are overloaded in N and suffer imbalances of Mg, K and P relative to N (Van Dijk and Roelofs, 1988).

The storage and retention of N by ecosystems is linked to the C/N ratio of the forest floor and mineral soil. Wide ratios encourage microbial immobilization. Nitrogen accumulation ceases when C/N ratios fall below ~ 20:1. In some soils, if N accumulates in excess of plant and microbial needs, it is nitrified and nitrate leaching occurs (Foster et al., in press). If microbial nitrification is limited under acidic soil conditions and under conditions of greatly elevated atmospheric N inputs, as in parts of central Europe, Schulze (1989) has hypothesized that trees may preferentially take up ammonium rather than nitrate with an antagonizing effect on the uptake of Mg. Simply put, atmospheric N promotes growth, thereby increasing the tree's demand for nutrient cations such as Mg, Ca and K.

Many boreal forests and some temperate ones are generally considered N limited and are likely to respond positively to additional atmospheric N input. Average rooting zone C/N ratio's presented in Tables 5.4.1 and 5.4.2 suggest that atmospheric N will be retained within the soil on most coniferous and some hardwood sites. Considerable N must accumulate in soil before a narrowing of the ratio would be effected. It is unlikely, therefore, that current rates of N deposition in Canada will produce nutrient imbalances in the forest vegetation in the short-term. However long-term impacts should be evaluated by empirical experimentation and application of nutrient cycling simulation models.

5.4.1.2.5 Terrestrial and Aquatic Linkages

Terrestrial and aquatic ecosystems are linked by the hydrologic cycle. The degree of influence of terrestrial drainage waters on streams and lakes depends on the ratio of incident precipitation falling directly on the water body; the amount contributed by terrestrial runoff; and the inflow from other water bodies in the system. Jeffries et al. (1988b) report 9-13% of precipitation falls directly on lake surfaces in the Turkey Lakes Watershed (TLW). Also, as might be expected from a lake chain system such as the TLW, drainage from the terrestrial basin immediately surrounding the lakes is a more important input for a headwater lake (58%) than for lower lakes (20-28%). Water input from the next higher lake becomes the predominating source (64-73%) for the lower lakes. In-stream processes such as sediment denitrification (Swank and Caskey, 1982) and phosphorus spiralling (Mulholland et al., 1985) may

also influence streamwater chemistry while in-lake processes can alter terrestrial inputs before they are passed on down the system.

Nicolson (1988) emphasized that care must be taken in the scale and definition of 'headwater' basins because, as demonstrated, the water and chemical signature of each basin can be unique, depending on groundwater contribution, channel length and morphology, vegetation type, soil depth, geologic make-up of the bedrock and overlying tills, degree of soil development and topographic position.

The routing of water along surface and subsurface flow paths within catchments is important because it influences the timing of streamflow and the chemical composition of water reaching the streams (Bricker, 1987). In order to understand both the hydrology and the chemistry of the water in catchments, it is necessary to define the actual pathways along which water moves through these systems. Surface runoff is not a major flow path and seldom, if ever, occurs in undisturbed forested basins (Hewlett, 1982). Sub-surface flow delivers most of the streamwater under snowmelt and flood conditions (Hewlett and Hibbert, 1963; Whipkey, 1969). Isotopic and hydrochemical studies show that, in most cases, the bulk of water in storm runoff is old water that has had a significant period of residence in the soil or groundwater system (Kennedy et al., 1986; Pearce et al., 1986; Sklash et al., 1986).

On the Canadian Shield, 40-90% of peak stream runoff may be composed of water that has passed through the saturated groundwater zone (Bottomley et al., 1984).

Nicolson et al. (1987) show one stream to be more sensitive to acidic inputs because it received water dominated by Ca^{2+} and SO_4^{2-} from the unsaturated and shallow groundwater zones while the second stream received more input of Ca^{2+} and HCO_3^- dominated water from the deeper groundwater.

Chemically, approximately 70% of the Ca^{2+}; 65% of the Mg^{2+}, Na^+, K^+, SO_4^{2-}, Cl^- and NO_3^-; but only 15% of the NH_4^+ and 25% of the H^+ are delivered to a 'headwater' lake from terrestrial sources (Jeffries et al., 1988). These figures generally drop to the 20-50% (with H^+ down to 1-2%) range for lower lakes in the chain, where upper lake inputs assume a more dominant role.

The detailed routes through which water travels in catchments are poorly known and there is a need to combine physical, chemical and isotopic methods to define flow paths, residence times, contact surfaces and solute sources for runoff from terrestrial catchments. With this information, uncertainties related to the transport of ions from the atmosphere to surface waters via terrestrial runoff can be resolved.

5.4.1.3 CANADIAN DECLINE PROBLEMS

Unlike the forest decline currently occurring in Europe, Scandinavia and the northeastern United States, where the problem has been primarily associated with the coniferous forests, forest decline in Canada has been historically and is currently

a phenomenon associated with the deciduous forest. Eastern Canada has experienced declines and diebacks of yellow and white birch, sugar maple, white and green ash, oak, elm and chestnut. With the exception of elm and chestnut, which have been extirpated across most of their former range as a result of introduced species-specific pathogens, these periodic tree declines have varied substantially in severity, extent and temporal distribution.

There is general agreement that the current episode of forest decline, particularly the sugar maple component, is both more severe and extensive now than has occurred in the past, and that exposure to atmospheric pollutants is a contributing factor. However, it is difficult to determine if this is the case for two main reasons. First, quantitative forest health surveys have not been consistently conducted on a regional basis. Therefore, the historical database does not exist with which comparisons can be made to the current situation. Secondly, the cause of forest decline at a particular time and location is almost certainly multi-factorial. Natural and anthropogenic stresses may act cumulatively or synergistically on the forest ecosystem to precipitate a decline episode. Direct cause and effect relationships are, therefore, elusive.

5.4.1.3.1 Sugar Maple Decline : Historical Perspective

Sugar maple decline is not a recent phenomenon, nor is it restricted geographically. It was first reported in 1913 in Pennsylvania and New Jersey (Hartley and Merriel, 1915). Subsequent to several episodes in the northeast U.S. (1913 to 1930), the first reported Canadian incidence of maple decline was in 1931-32 in Beauce County, Quebec (Forestry Canada, 1962). Maple decline appeared in Ontario in 1947-48 (Nordin, 1954). Many occurrences were noted in the Lake States and Ontario throughout the 1950's and 1960's (Skilling, 1959; Forestry Canada, 1960,; Griffin, 1965). As recently as 1980, the FIDS monitored maple decline in Ontario on 25,000 ha in Algonquin Provincial Park and 8,000 ha in Parry Sound. Tree mortality exceeded 25% in these areas (Forestry Canada, 1980).

Decline episodes appear to be increasing. For example, of 102 episodes in Ontario, 65 were recorded after 1975 (McIlveen et al., 1986). Whether this is a true increase in decline or simply an increase in the frequency of reporting is, however, not clear. Currently, maple decline is a concern in scattered locations across Ontario and New Brunswick. The most severe and extensive decline in recent time is occurring in a large area mostly south of Quebec City, Quebec.

5.4.1.3.2 Current Status of Sugar Maple Decline: Quebec

Quebec produces over 70% of the world's supply of maple syrup. With such a large market share, the syrup producers are keenly aware of changes in their sugar bushes. Maple syrup producers identified 1978 as the first year of the current

episode of sugar maple decline in this province. The area affected increased significantly in 1981. By 1982, decline symptoms were evident across almost 2,000 km^2 of sugar maple forest in Quebec (Roy et al., 1985). A multidisciplinary research team was formed in the fall of 1982 to determine the extent of the problem, identify the cause(s), and develop mitigative measures.

Several surveys were conducted to address the first objective. In 1982, the Ministère de l'Agriculture, des Pêcheries et de l'Alimentation sent a questionnaire to 7,000 maple syrup producers. Based on 2000 survey returns, the data indicated that, although decline symptoms were observed by the syrup producers throughout the province, the problem was concentrated in three regions: Beauce, Amiante and Bois Francs. A subsequent ground survey conducted in 1983 by Forestry Canada corroborated the geographic distribution of maple decline in Quebec (Lachance, 1985).

Survey activity continued between 1985 and 1987. During this time the Quebec Ministère de l'Energie et des Ressources conducted a forest health assessment survey from an aerial platform of all maple stands within the sugar maple range in Quebec. This survey effort covered 2,119,252 ha of sugar maple stands. The results (**Table 5.4.3**), revealed that about 50% of the stand area exhibited only marginal decline symptoms (<10% defoliation.). About 47% of the stand area fell into the light decline category (11% to 25% defoliation), and the remaining 3% of the stand area were moderately to severely affected (>25% defoliation) (Bordeleau, 1987).

Table 5.4.3

Condition of Maple Stands in Eight Regions Included in the 1985, 1986 and 1987 Aerial Surveys.

Region	Decline Status								Total
	Defoliation <10% Healthy-Light Decline		Defoliation 11% - 25% Light Decline		Defoliation 26% - 50% Moderate Decline		Defoliation >50% High Decline		
	ha	%	ha	%	ha	%	ha	%	ha
Bas Saint-Laurent, Gaspesie	90,136	80.6	17,580	15.7	3,367	3.0	744	0.7	111,827
Saquenay, Lac Saint Jean	6,125	62.7	3,575	36.6	69	0.7	-	-	9,769
Quebec	130,752	39.8	172,197	52.4	22,981	7.0	2,473	0.8	328,403
Trois Rivieres	49,890	34.3	88,893	61.2	5,823	4.0	745	0.5	145,351
Estrie	155,738	68.1	66,275	29.0	5,036	2.2	1,628	0.7	228,677
Montreal	272,603	41.7	370,112	56.6	10,713	1.6	841	0.1	654,269
Outaouais	322,939	55.4	254,209	43.6	5,495	0.9	131	0.1	582,774
Abitibi	30,306	52.1	27,538	47.3	338	0.6	-	-	58,182
Total	1,058,489	50.0	1,000,379	47.2	53,822	2.5	6,562	0.3	2,119,252

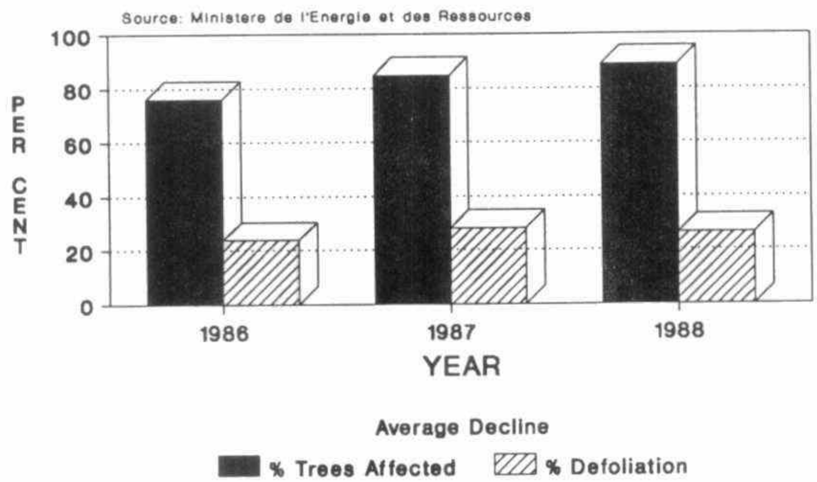
Source: Bordeleau, 1987

Quebec researchers established a network of 256 study plots to address the second and third objectives (causal factors and mitigative measures). These plots were stratified by plant communities and associated ecological characteristics as well as by the intensity and type of maple syrup production. For all plots, the frequency of decline increased from 76.0% of the trees exhibiting symptoms in 1986 to 88.8% in 1988. However, the severity of decline has now decreased marginally from an average of 28.1% defoliation in each tree in the peak year (1987), to 26.5% in 1988 (**Figure 5.4.1**).

In the Appalachian region, where the decline was concluded to be the most severe and extensive, a survey of 133 plots revealed that maple stands have continued to deteriorate. In this region, the frequency of trees with decline symptoms increased from 71.2% in 1984 to 91.3% in 1988. In the same period, the average severity of decline increased from 20.7% in 1984, peaked at 32.5% in 1987, and fell marginally to 30.2% in 1988 (**Figure 5.4.2**).

The decline of sugar maple stands in Quebec is most severe and extensive in the

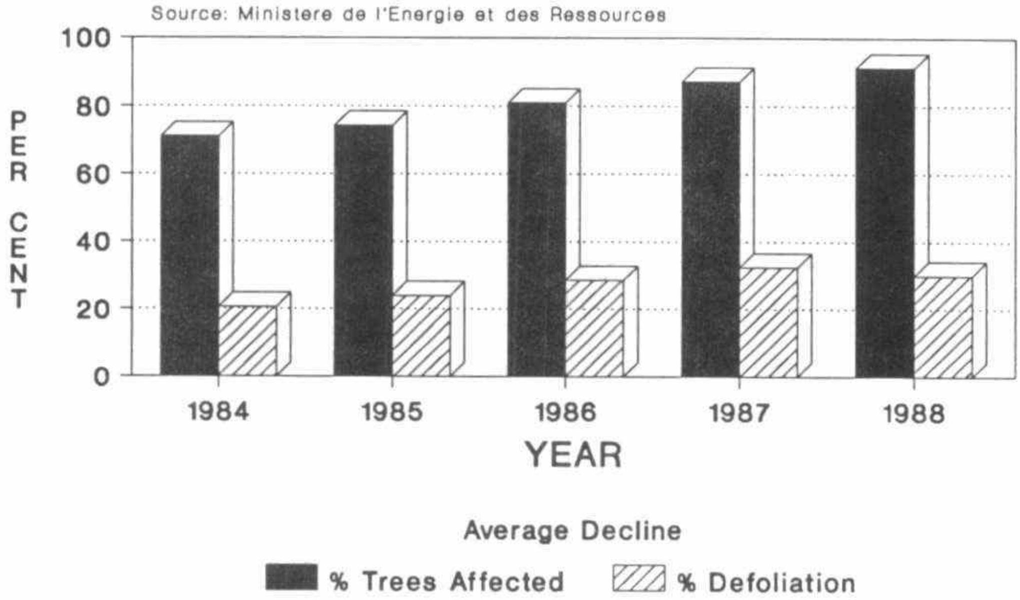
Fig. 5.4.1 Maple Decline in Quebec
256 Study Plots, Province-wide



area south of Quebec City. This area experienced severe defoliation by forest tent caterpillar in 1980 and 1981. The winter of 1980/81 was characterized by periods of intense cold and very low snowfall plus an unusual thaw in February during which a commercial flow of sap occurred. The thaw was followed by a deep freeze. Growing season droughts occurred in both 1982 and 1983. Within this and other areas where decline has been observed, it is more prevalent on thin soils, excessively dry or wet sites, and in stands which have been heavily thinned. Although these are unquestionably significant stresses, Quebec researchers conclude that they are contributing, rather than inciting causal agents in the current decline phenomenon because decline is occurring elsewhere in the province in areas where these stresses have not occurred.

The area south of Quebec City, where the decline is most severe and extensive also receives the highest rate of atmospheric pollutants in the province. The soil at many of these sites appears to have developed magnesium, potassium and possibly phosphorous deficiencies (Bernier and Brazeau, 1988 a,b). For these reasons, atmospheric pollutants are strongly suspected as contributing to the current decline phenomenon.

Fig. 5.4.2 Maple Decline in Quebec
Appalachian Region (133 Plots)



5.4.1.3.3 Current Status of Sugar Maple Decline: Ontario

Sugar maple decline in Ontario has been assessed in three ways. First, information was gathered in 1984 from all Ministry of Natural Resources (MNR) districts on the extent and severity of maple decline. Second, in 1985 a survey was distributed to all members of the Ontario Maple Syrup Producers Association (OMSPA). Third, as a response to information received from the MNR and concerns expressed by the OMSPA, the Ministry of the Environment (MOE) conducted an extensive province-wide hardwood forest decline survey in 1986. This survey was repeated in 1987 and 1989 to evaluate spatial and temporal decline gradients. In addition the etiology of sugar maple decline was assessed at 11 sites from 1984 through 1989.

Based on the MNR survey, maple decline started in the late 1970s and most

commonly occurred in isolated pockets. Decline was most frequent in areas of recent insect defoliation and in woodlots with poor management regimes; however, many cases of "unknown causes" were also reported.

One third of the OMSPA survey respondents reported that hardwood decline was currently (1985) a problem in their woodlot. Of the 33% reporting decline, 72% said it was getting worse, and 89% said they had not experienced a similar decline in their woodlot. The most frequently cited year in which symptoms first became apparent was 1982 (20%), followed by 1980 (17%) (McLaughlin and Butler, 1987).

The OMSPA survey data indicated that maple decline in sugar bushes was most frequent in areas adjacent to Georgian Bay, Algonquin Park, the Ottawa valley, and in counties in southwest Ontario. These results are in general agreement with information obtained from the MNR offices.

The province-wide hardwood forest decline survey which was initiated by the MOE in 1985, was completed in 1986, and repeated in 1987 and again in 1989. In total, 110 permanent forest observation plots were established across the range of sugar maple in Ontario. Each plot contained 100 trees greater than 10 cm in diameter, for a total of 11,000 surveyed trees. Plots were not located in woodlots managed for maple syrup production.

Decline status of the trees was determined with a crown condition assessment developed specifically for the survey (McLaughlin et al., 1988). The assessment procedure combines field observations of branch dieback and foliar colour and size into a numeric value referred to as a Decline Index (DI). The DI ranges from 0, a healthy tree with no symptoms, to 100, which represents a dead tree. Sugar maple was the target species, comprising 75% of all trees tallied. Tree condition data were also collected for 21 other species which occurred on the plots, although only 9 species each comprised more than 1% of the survey sample population. **Table 5.4.4** summarizes the mean Decline Index (DI) for these 9 tree species.

Table 5.4.4
Relative Condition of Nine Hardwood Tree Species in Ontario

Tree Species	% of Survey Total	Mean Decline Index
Sugar Maple	74.8	12
American Beech	3.0	13
White Ash	3.7	17
Basswood	3.0	18
Yellow Birch	1.7	20
Red Oak	1.5	20
Red Maple	3.1	22
White Birch	1.0	24
Black Cherry	1.6	28

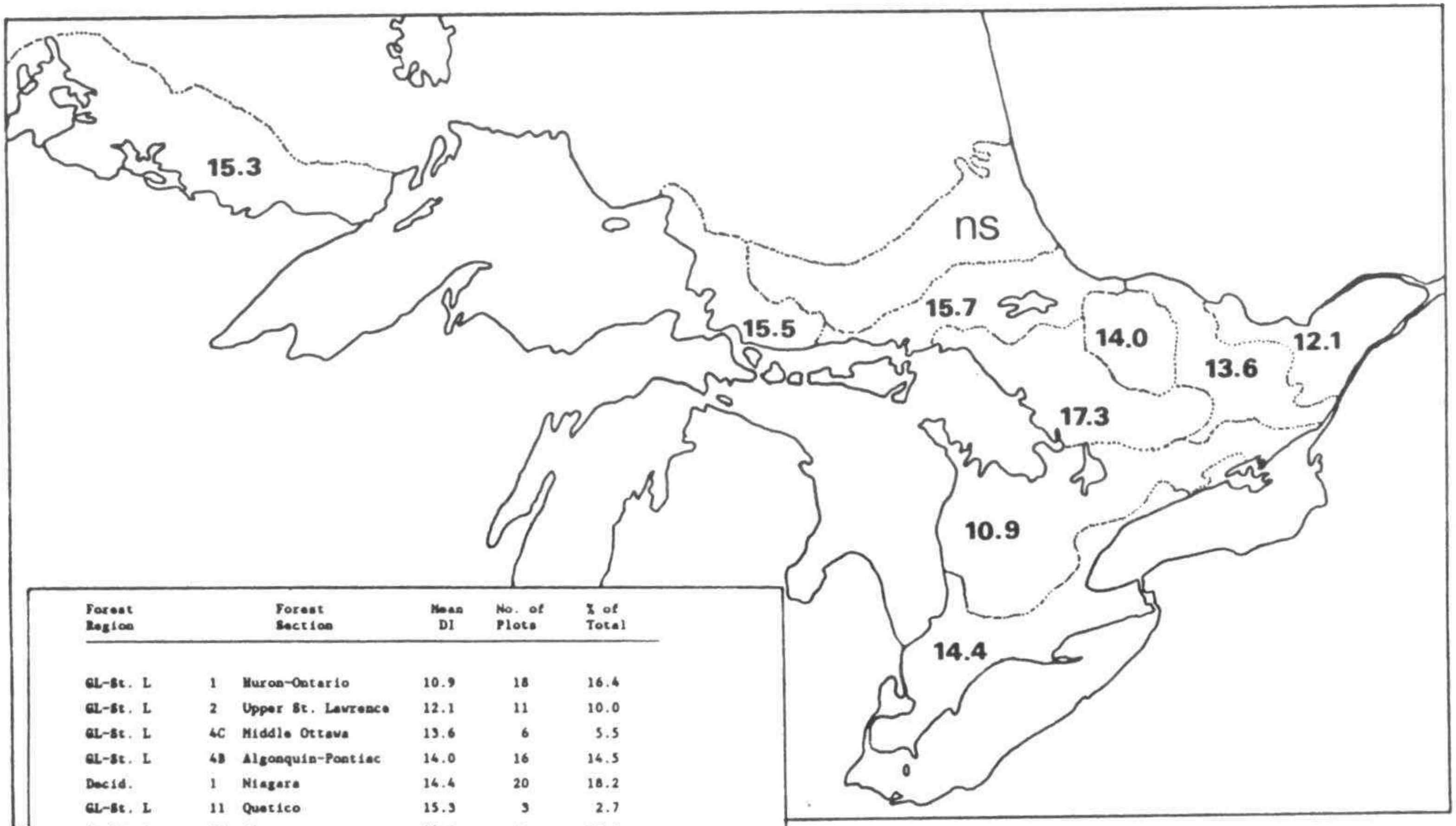
Based on survey of 11,000 trees in 1986. Only species comprising more than 1% of the tally each are listed in the table. Data adapted from McIlveen et al. (1988).

Decline Index Gradient: (0 = no symptoms, 100 = dead tree)
0-5 = healthy
6-10 = trace to light decline
11-20 = light to moderate decline
21-30 = moderate decline
31-40 = moderate to severe decline
41-50 = severe decline
>50 = very severe decline

These data suggest that additional hardwood species are experiencing decline, and that sugar maple is in better condition than many other hardwood trees in Ontario.

Figure 5.4.3 illustrates the mean 1986 DI for each of the nine forest sections (Rowe, 1972) in Ontario. The mean DI is highest in the area between Georgian Bay and Algonquin Park, and across the northern edge of the hardwood forest range. The southwest of the province is also an area of (relatively) high decline. In contrast, decline indices are lowest across south central and southeast Ontario. At $p<0.01$, the mean DI must be more than 4.7, and at $p<0.05$, the DI difference must be at least 3.0 to be statistically significant. These statistics illustrate that there are regional differences in the condition of the hardwood forest resource in Ontario.

Data from the 1987 repeat survey revealed that tree condition improved (relative to 1986) at 6% of the plots, and deteriorated at 29% of the plots. The remaining 65% of the survey plots did not change significantly (change identified as the 1987 plot mean DI greater or less than one



Forest Region		Forest Section	Mean DI	No. of Plots	% of Total
GL-St. L	1	Huron-Ontario	10.9	18	16.4
GL-St. L	2	Upper St. Lawrence	12.1	11	10.0
GL-St. L	4C	Middle Ottawa	13.6	6	5.5
GL-St. L	4B	Algonquin-Pontiac	14.0	16	14.5
Decid.	1	Niagara	14.4	20	18.2
GL-St. L	11	Quetico	15.3	3	2.7
GL-St. L	10	Algoma	15.5	11	10.0
GL-St. L	4E	Sudbury-North Bay	15.7	13	11.8
GL-St. L	4D	Georgian Bay	17.3	12	10.9
Least Significant Difference (at <0.01)			4.7		

forest classification after Rowe, 1972
ns region not surveyed

Fig. 5.4.3

HARDWOOD TREE CONDITION IN ONTARIO
1986

standard deviation of the 1987 survey mean, relative to the 1986 plot data). Moderate to severe insect defoliation in 1987 occurred at about one half of the 29% of the plots in which tree condition deteriorated. However, defoliation was not a factor at the other survey plots which had declined.

The fixed plot survey data confirmed observations by the MNR and the OMSPA that, in the 1980s, maple decline was a scattered, isolated phenomenon in Ontario, although regional differences were apparent. The geographic distribution of maple decline symptoms in Ontario correlates only marginally with atmospheric pollution. Decline is prevalent in the southwest where pollution levels are highest (>35 kg/ha/yr wet sulphate and (7 hour) growing season ozone concentration which averages 50 ppb). Higher rates of decline occur in the central and northern regions where pollution levels are lower. However, trees in these areas are considered more sensitive to potential indirect pollution effects because of the additional stress associated with survival on shallow, poorly buffered soils of the Precambrian Shield.

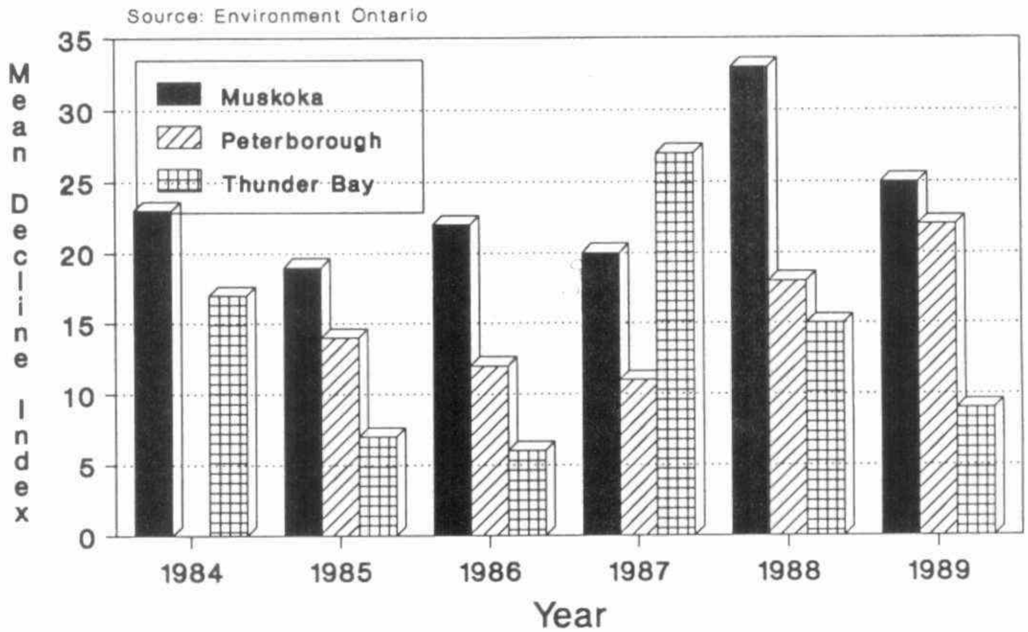
The etiology of sugar maple decline was extensively examined at eleven selected study sites from 1984 to 1989. The sites were chosen to reflect a gradient of decline and management regimes. Tree condition, as rated by the DI method, did not change significantly at the study sites from 1984 to 1986. Tree condition deteriorated in 1987 and again in 1988, following severe defoliation by forest tent caterpillar and two consecutive unusually warm and dry growing seasons. A moderate improvement in average tree condition occurred in 1989 (See **Figure 5.4.4**). The causes of maple decline at these intensively studied sites was determined to be multifactorial and variable among sites. Severe defoliation by forest tent caterpillar from 1975 to 1978 was the prime inciting factor. This occurred concurrently with spring droughts in 1976 and 1977. Climatic stress events continued through to and including 1988, and in conjunction with Armillaria root rot, were important contributing factors. Poor woodlot management was a factor only on a few sites.

Atmospheric pollution can not be eliminated as either a predisposing or possibly even a contributing causal factor. Soil-available Al levels ranged from 10 to 40 ppm at the poorly buffered sites on the Precambrian Shield. Fine-root Al concentrations of declining trees were consistently 50% to 100% higher than in roots from healthy trees. Although it was considered possible that the elevated aluminum levels were an effect rather than a cause of decline, the authors ruled this out based on the fact that other soil ions did not undergo a similar type of passive adsorption onto the injured root tissues.

Tree growth, measured as annual growth rings, has been declining at most of the intensive study sites since about 1960. Growth reductions were greatest in trees exhibiting decline symptoms and in areas receiving the highest pollution loadings (12.2% to 17.9% narrower annual rings, long term average after 1945, see **Table 5.4.5** - McLaughlin et al., 1989).

Maple Decline in Ontario

Fig. 5.4.4 Change in Tree Condition



Historical growth trends of sugar maple throughout Ontario have been under investigation since 1986. To date over 1000 increment cores have been collected and 85 mature trees have been destructively sampled for detailed stem analysis. These data were collected from stands under a variety of management regimes and site conditions. Stands exhibiting decline symptoms were avoided. Whereas stem analysis was conducted only on mature trees, increment cores were collected from all trees over 10 cm DBH encountered in the growth study plots. The time periods for comparison of long term growth trends were selected to reflect conservative assumptions regarding historical air quality. The "pre high pollution" period was considered as 1900 to 1930. Although air pollution associated with industrial and residential fossil fuel consumption was certainly a factor in this time period, it was

Table 5.4.5

Average Annual Ring Values for Sugar Maple Pre/Post 1945 from Three Atmospheric Pollution Deposition Zones in Ontario

Study Region (n)	Mean Annual Ring Value* (SD) 1905 - 1944	1945 - 1984	% Difference	Atmospheric Pollution Loading SO_4^{**}	NO_2^{**}	O_3^{***}	pH
Thunder Bay (1)	1.15 (0.31)	1.08 (0.28)	-6.1 (ns)	12	3	<20	4.8
Muskoka (8)	1.23 (0.15)	1.08 (0.18)	-12.3 (p<0.01)	34	9	40	4.2
Peterborough (2)	1.23 (0.28)	1.01 (0.27)	-17.9 (p<0.01)	43	8	50	4.2

data adapted from McLaughlin et al., 1989

* Indexed Ring Value, after Fritts, 1966

** Total wet and dry deposition, kg/ha/yr, mean 1981 - 1984, from Tang et al., 1986

*** 7 hr daylight average (000-1600 EST) June to August, ppb, mean 1974 - 1981, from Linzon et al., 1984

% difference is 1945-1984 less than 1905-1944

accepted that regional impacts would be marginal relative to recent air quality. By comparison, the period from 1955 to 1985 was selected as the time frame in which regional air quality would be most severely impacted. The increment core data were stratified into three relative air pollution deposition zones (i.e., High, Moderate and Low). **Table 5.4.6** summarizes the long term mean annual incremental growth for outwardly healthy sugar maple in Ontario for these two time periods across a pollution gradient.

In the highest deposition zone, sugar maple growth has averaged 27% less in the most recent 30 year period compared to the first 30 years of this century, when regional air pollution levels were substantially lower. Tree growth has averaged more than a 6% reduction in the moderate pollution loading zone and has improved more than 39% in trees growing in the area receiving the lowest deposition in the same time period.

Table 5.4.6

Mean Annual Ring Width for Two 30-Year Periods in Three Air Pollution Deposition Zones in Ontario

Zone	Mean Ring Width - mm (SD) 1900 - 1939	Mean Ring Width - mm (SD) 1955 - 1985	% Diff.	p<	Total Deposition $SO_4^{2-} + NO_3^-$	Ozone (ppb)
High	1.92 (0.17)	1.40 (0.17)	-27.1	.001	60	50
Moderate	1.10 (0.13)	1.03 (0.17)	-6.4	.05	42	30
Low	0.78 (0.10)	1.09 (0.13)	+39.8	.01	24	20

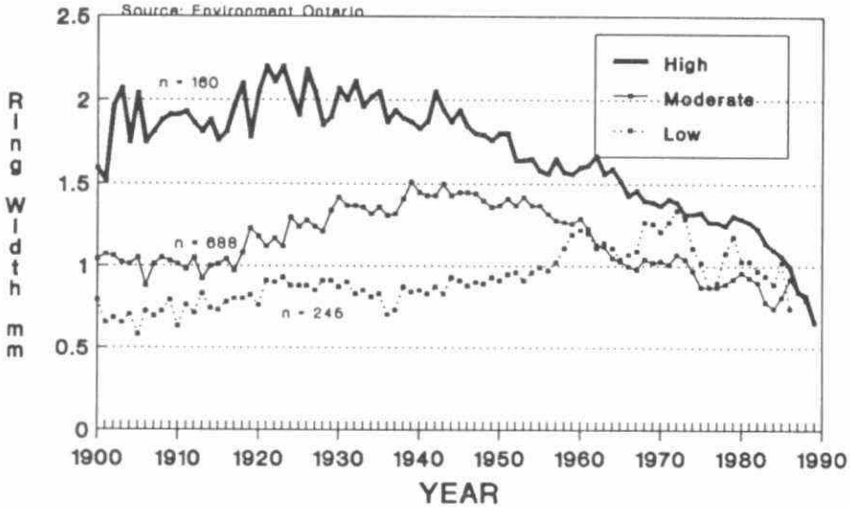
Total SO_4^{2-} and NO_3^- are kg/ha/yr (SO_4^{2-} +N), wet and dry deposition.
O_3 is 7-hr (0900-1600 E.S.T.) growing season mean (June-August) (Avg. 1974-1981).

It is evident in **Figure 5.4.5**, which illustrates the sugar maple growth chronology derived from increment core data, that there has been a marked tendency towards a reduction in absolute growth in the sampled trees beginning between about 1945-1955 in areas where regional air pollution is prevalent. These data are preliminary. Site specific factors and the relationship of various climatic parameters to changes in tree growth are currently being investigated. However, the consistency of the growth patterns among sites within each area and the similarity of the long term trends strongly suggests a regional abiotic stress as a causal factor.

5.4.1.3.4 Recent Concerns Regarding White Birch Decline

Like sugar maple, white birch has experienced episodic decline in eastern Canada. With few exceptions, these historic declines have been associated with residual white birch stands subsequent to logging activity. White birch has a shallow root system which is very sensitive to environmental change associated with site disturbance. Many of these past declines were found to be related to subtle increases in soil temperature or alterations in moisture availability. Currently, white birch decline is evident in the Bay of Fundy area on Canada's eastern maritime coast and in the area around Wawa, Ontario, adjacent to Lake Superior. These current birch declines are both occurring in relatively undisturbed forest stands (i.e.

Fig. 5.4.5 Sugar Maple Growth Across Three Pollution Gradients in Ontario



no recent logging) and in areas which experience frequent marine fog.

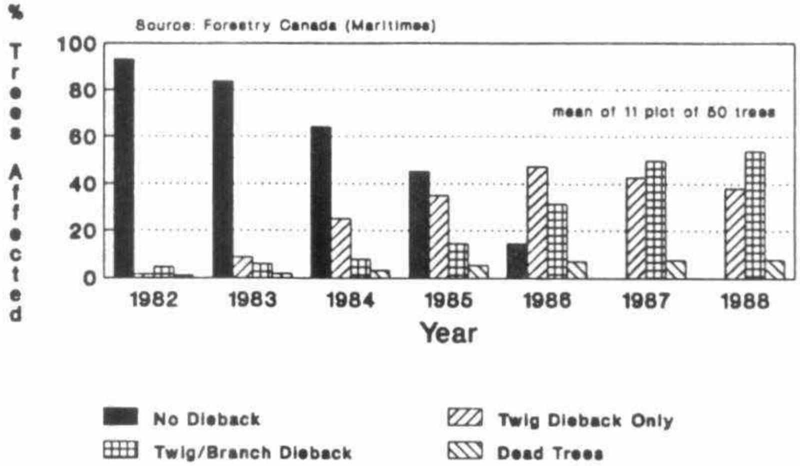
Bay of Fundy White Birch Decline

The deterioration of white birch adjacent to the Bay of Fundy has been noted by the FIDS annual surveys since 1979 (Magasi, 1989). This provided an opportunity for Forestry Canada to study populations of two closely related birch taxa, Betula papyrifera Marsh. and B. cordifolia, which are in early stages of a progressive decline. A consistent increase in the frequency of decline symptoms and tree mortality has been observed since 1982 (**Figure 5.4.6**).

The symptoms are not readily explainable by insect and disease associated with the stands (Magasi, 1989). Air pollution was suspected due to the relatively high wet sulphate deposition in the area (>20 Kg/ha/yr). Furthermore, the reporting of low pH's in marine fog from the Coast of Maine (Weathers et al., 1986) and the overlap of the affected area of birch with the area intercepting sea fog, lead to the suspicion that this form of deposition was involved. Elevated ozone has also been reported along the Maine coast and southern New Brunswick (Jagels, 1986; Tims and Knight, 1985).

Fundy White Birch Condition 1982 - 1988

Fig. 5.4.6



Study sites were selected along a transect through the affected area north from Point Lepreau. Sea fog chemistry and the initiation, development and intensity of birch browning symptoms were monitored at each site. Results from 1986 and 1987 confirmed that sea fog averaged pH 3.6, which is one pH unit lower rainfall in the area (Cox et al., 1989). Ozone levels were higher nearer to the coast than those in urban St. John. Using a simple model (Unsworth and Crossley, 1985; Cox et al., 1989), it was found that the deposition of acidity by sea fog to the affected birch stands was as important as rain. This form of deposition was not detected by conventional collectors in wet deposition networks and may indicate a serious underestimate of wet deposited acidicity. The proportion of fog acidity contributed by dry deposition is unknown. However, dry deposition alone would not account for the additional acidity produced by the unique chemical environment of marine fog. (Eatough et al., 1984; Miller et al., 1987).

Cox et al. (1989) summarize the correlation between the intensity of birch leaf injury at each of five sites across the affected area with various air quality parameters monitored in 1987.

The preliminary data indicate significant statistical relationships between intensity of birch browning and volume weighted H^+ and NO_3^- concentrations. Foliar browning

was also found to increase with leaf age, suggesting cumulative damage and possible predisposition of the foliage to leaf spot fungus (Septoria betulina), a pathogen occasionally associated with foliar damage.

The role of wet deposition via fog in causing direct foliar damage may differ substantially from that of rain. With fog, acidic pollutants may accumulate on leaf surfaces to much higher concentrations and may have a longer residence before being washed off (Unsworth and Crossley, 1985). These deposits may directly damage plant surfaces such as cuticular membranes, leading to dehydration and necrosis of underlying tissue.

The high frequency of mountain paper birch (Betula cordifolia) close to the coast may contribute to overall stand sensitivity. This species is at the southern limit of its range at sea level. Mountain paper birch prefers a cool and moist habitat (Rosendhal, 1928), which is provided by high summer fog frequency. It is hypothesized that repeated incidence of leaf browning and early foliar senescence may be associated with fog acidity and may be contributing to the initial deterioration of these stands.

Wawa, Ontario White Birch Decline

The transitional forest along the shoreline of Lake Superior in the area around Wawa, Ontario experiences frequent summer marine fog. A progressive decline of white birch in this area has been observed since the early 1970's. The decline was at first assumed to be related to SO_2 emissions from the iron ore sintering plant in Wawa. However, the extent of decline was inconsistent with the well-defined plume zone from this source, but was consistent with areas receiving marine fog.

The Ontario Ministry of the Environment initiated birch decline studies in the Wawa area in 1984 to determine if atmospheric pollutants, including fog-acidity, were causal factors. Sites were selected and foliar, root, sap, and xylem samples were collected from symptomatic trees. Fog sampling also was conducted. Only preliminary results are available at this time. Soils are generally coarse textured, poorly buffered, and naturally acidic, with pH extremes as low as pH 2.9 (surface horizon). Fog was acidic (pH 3.6 to 3.9)although only a few samples could be collected due to unusual fog-free conditions during 1988 and 1989. Electrophoresis of active bud tissue confirmed that Betula cordifolia comprised the majority of the older trees in the stands, whereas most of the regeneration and younger trees were Betula papryifera. Tree decline was more frequent among B. cordifolia. It is hypothesized that subtle climate change may be gradually altering the site characteristics to disfavour B. cordifolia. As B. cordifolia slowly disappears from the stand it is being replaced by the less site-demanding B. papryifera. This transition process and the selection against the less-tolerant B. cordifolia may be hastened by acidity-related environmental stress.

5.4.1.3.5 Possible Future Problems

The forest is a dynamic ecosystem. Forest decline may be a periodic or cyclical event which occurs in response to specific environmental stresses or to accommodate natural forest succession. Now, however, in addition to the many natural stresses, the forest must also endure the cumulative, or perhaps synergistic, effect of anthropogenic stress imposed by atmospheric pollutants. At times when the forest is naturally stressed, the additional pollution stress may be significant enough to initiate a local or regional decline.

As mentioned earlier, tree mortality can rarely be directly related to exposure to LRTAP. Usually the tree dies of a secondary pathogen, thereby obscuring the pollution connection. As a result, in an ecosystem as complex as the hardwood forest, it is unlikely that the contribution of long range transported air pollution to a specific decline episode can ever be confidently and quantitatively identified.

Further complicating this situation is the potential impact of climate change. Although the magnitude and timing of climate change are still contested, there is general agreement that the driving force behind increasing levels of greenhouse gases is atmospheric pollution, particularly those associated with the combustion of fossil fuels. Again the connection between tree decline and pollution is obscured. The trees may die from secondary infection as a result of reduced vigour, caused by unfavourable climatic events, which are initiated by higher levels of atmospheric greenhouse gases, which in turn are directly related to air pollution.

5.4.2 AGRICULTURE

5.4.2.1 ACIDIC PRECIPITATION

Results of studies in the scientific literature indicate that repeated occurrences of highly acidic rain or fog with pH<3.0 will increase the likelihood of foliar injury to agricultural crops. However, with one exception, simulated acidic rain (SAR) experiments in the field have not been able to prove that ambient or near-ambient rain acidity reduces crop yield. It has been shown that the yield of "Amsoy 71" soybean exposed to SAR with pH 4.1 was reduced significantly. However, no negative yield effects have been observed in any other soybean cultivars investigated in a number of studies conducted in the U.S.A. and Ontario (**Table 5.4.7**).

Table 5.4.7 **Soybean cultivars investigated in SAR studies**[a]

Cultivar	References
AMSOY 71	Evans and Curry, 1979; Evans and Lewin, 1981; Evans et al., 1980; 1981; 1983; 1984a; 1984b; 1985; 1986; Irving et al.,1984; Porter et al.,1987
ASGROW	Evans et al., 1986
BEESON	Evans et al., 1984b; Jacobson, 1980; Troiano et al., 1982; 1983
CLARK	Keever and Jacobson, 1983
CORSOY	Evans et al., 1986
DAVIS	Brewer and Heagle, 1983; Heagle et al., 1983; Johnston and Shriner, 1985; Norby and Luxmoore, 1983; Norby et al., 1985
EVANS	Cohen et al., 1981; Lee et al., 1981
FORREST	DuBay and Heagle, 1987
HARK	Cohen et al., 1981; Lee et al., 1981
HOBBIT	Evans et al., 1986
HODGSON	Kuja and Dixon, 1989
LEE	Shriner and Johnston, 1981
NORMAN	Cohen et al., 1981; Lee et al., 1981
OR-10	Cohen et al., 1981; Lee et al., 1981
WELLS	Evans et al., 1984b; Irving and Sowinski, 1981; Irving and Miller, 1981
WILLIAMS	Craker and Bernsteirn, 1984; Evans et al., 1984b; Troiano et al., 1982; 1983
WILLIAMS 82	Porter et al., 1987

[a]from Porter et al., 1987 and more recent studies.

Also, studies with soybean (Takemoto et al., 1987; Norby et al., 1985; Brewer and Heagle, 1983) have failed to show significant interactive effects between acidic rain and ozone.

The fact remains that the available scientific information has not established any measurable or consistent crop yield response from the direct effects of natural ambient levels of rain acidity (Irving, 1988). However, the following caveats should be noted.

Dose-response field studies have been performed on only a small number of crop species; e.g. soybeans, beans (snap), clover, corn, oats, potato, radish, ryegrass and tobacco (Troiano et al., 1984); Menchaca and Hornung, 1989; Banwart, 1987; Pell and Puente, 1987; Pell et al., 1987; Evans et al., 1982; Troiano et al., 1982a,b; Rathier and Frink, 1984). For most crops tested, screening experiments have been conducted on only one or two cultivars to determine species sensitivity to rain acidity (Irving, 1983). Studies have demonstrated that plant response to SAR may be not only species dependent but also strongly cultivar dependent. Also, perennial crop groups such as fruit trees and forage crops have not been examined adequately (Kuja, 1988).

There is recent experimental evidence that sexual reproduction and seed set in corn (*Zea mays*), may be adversely affected by increased rain acidity (DuBay, 1988; 1989). In other studies, pollen germination was significantly reduced on Pioneer 3747 corn silks exposed to one episode of simulated acidic rain of pH 4.6, 3.6 and 2.6, compared with germination on silks treated with rain at pH 5.6 (Wertheim and Craker, 1987; 1988). Further research is warranted to determine if acidic rain at pH levels recorded for individual events in North America could inhibit plant reproductive processes critical to seed set in various field crops.

Although there are only a few studies to date, the crop leaf cuticle may be another plant component which is sensitive to perturbation by simulated acidic rain and other air pollutants. Wax production in ryegrass was initially found to be stimulated by SO_2 fumigation (Koziol and Cowling, 1981). These effects were later found to be dependent upon genotype, SO_2 concentration and season (Shelvey and Koziol, 1986). Waxy leaves exhibit the most foliar injury while all four crops tested by Percy and Baker (1988) manifested changes in wax production at simulated acidic rain pH's <4.6. Wax composition was also affected as was wax crystal regeneration, which was less under the influence of rain at pH <4.2. These alterations to wax physio-chemical characteristics resulted in consequential effects on leaf wettability, rain retention and uptake of inorganic ions (Percy and Baker, 1988).

Given the limitations of experimental designs, there are factors inherent in all studies which influence results and must be recognized when interpreting observations from simulated acidic rain experiments. For example, it has been demonstrated that environmental conditions such as drought influence plant response to acidic precipitation (Banwart, 1988; Kuja and Dixon, 1989).

Environmental factors may account for conflicting results obtained in controlled environments vs. field situations and may partially explain temporal and site differences obtained in SAR field studies. Multiple combinations of chemical, physical and temporal variables of precipitation have not been examined, nor has the interactive role with dry acidic wet deposition in crop response been determined.

Finally, the effect of acidic wet deposition on most agricultural soils may be minimal because of relatively high buffering capacities. Modern farming practices, such as liming and return of crop residues may override depositional effects. However, the long term effects of acidic deposition on micronutrient cycling and plant availability in agricultural soils are not yet known.

5.4.2.2 OZONE

Plant response to elevated ozone exposure is based on a sequence of biochemical and physiological events which culminate in some type of injury expression. The resulting cellular disturbances involve changes in both functional and structural characteristics, resulting from disruption of cellular membranes. These disturbances can result in foliar pathologies, altered carbohydrate allocation, reduced growth and yield as well as impacts on plant communities and ecosystems (Guderian et al., 1985).

As with many other pollutants, ozone effects on plant foliage can be categorized into acute, chronic and subtle effects. Acute symptoms are characterized by bifacial foliar lesions while chronic symptoms develop more generally from pigmented lesions, stippling, and bleaching to general foliar chlorosis. Subtle effects include reductions in plant productivity or vigour without visible symptoms.

Exhaustive lists of crop sensitivity/resistance to ozone under short term, controlled environment conditions have been published (Guderian et al., 1985; Heck et al., 1977). From an ozone exposure perspective, comprehensive reviews of the available literature were made by Jacobson (1977) and Linzon et al., (1975) and concentration:time profiles for acute foliar injury were developed for sensitive, intermediate and less sensitive crop species. Based on these and other findings, Guderian et al. (1985) developed a set of maximum acceptable ozone concentrations which, if met, would provide reasonable protection of vegetation from foliar injury- producing, short term, acute exposures.

These concentrations are shown in **Table 5.4.8** below:

Table 5.4.8 Critical levels[1] for Plant Response (Foliar Injury) to Ozone

Exposure Duration (hr.)	Sensitive	Resistance Level Intermediate (ppb)	Less Sensitive
0.5	150	250	500
1.0	75	180	250
2.0	60	130	200
4.0	50	100	180

[1] from Guderian et al., 1985

In Canada, there have been several studies designed to assess foliar injury response to ambient ozone. These include New Brunswick (Tims, 1984; Tims and Knight, 1987), Ontario (Pearson, 1989), Quebec (Maltais and Archambault, 1985, 1986) and British Columbia (Runeckles, 1989). These programs have documented foliar injuries to a number of sensitive crops in New Brunswick (potato), Quebec (dry bean, soybean, tobacco), Ontario (dry bean, soybean, potato, tomato, onion, tobacco, cucumber, grapes, peanut) and British Columbia (pea, potato). Although none of these studies were designed to determine the exposure threshold for foliar injury, in all cases injuries were observed in locations where hourly ozone levels exceeded the Canadian objective at some point during the growing season.

In recent years, there has been a shift in research priorities from controlled environment systems to natural field exposure systems. The main reason for this shift was the growing body of evidence that indicated foliar injury was not an acceptable surrogate for ozone impacts on crop yield.

Any assessment of yield or quality parameters under field conditions is complicated by the ubiquity of ozone exposure, the effect of micro meteorological variables on ozone distribution within crop canopies and the effect of numerous biotic and abiotic factors which can alter plant response. Some of these difficulties have been partially overcome by refinement of field assessment techniques, including open-top chambers, open air fumigation systems and ambient air pollutant gradients (Ormrod et al., 1988).

While it is now recognized that indirect effects of air pollutant:biotic interactions have been underestimated and may be as important as the direct pollutant effects (Manning and Keane, 1988) there are few studies on the effect of disease epidemiology and crop yields. There are, however, numerous reports which demonstrate the potential importance of these interactions (Warrington, 1989; Houlden et al., in press)

The most comprehensive program which has emerged to address the issue of growing season ozone impact on crop yield is the National Crop Loss Assessment Network (NCLAN) in the U.S. This seven-year program was conducted from 1980-1986 at five geographic sites, chosen to represent distinctly different climatic conditions in regions growing different crop species. Open-top chambers were used to expose different agricultural crops to various regimes of ozone and sulphur dioxide, with plant yields being measured to determine dose:response correlations. The results of this program are described in numerous publications (Heck et al., 1982; 1983; 1984a; 1984b; 1988).

In the most recent NCLAN assessment of yield losses, a three-parameter, Weibull model (Rawlings and Cure, 1985) was used to predict production losses of major crops in the U.S. from 7 and 12-hour seasonal ozone mean concentrations of 40, 50 and 60 ppb. The results of these model predicted losses, which were based on NCLAN-style field exposures for 11 crops averaged 4.0, 7.6, and 10.1% for seasonal means of 40, 50 and 60 ppb, respectively (Heck et al., 1984a).

Utilizing the findings from the NCLAN experiments and other similar studies, many attempts have been made to find a better surrogate or descriptor for the exposure:response index or statistic that best characterizes crop response to ambient ozone exposure (Musselman et al., 1988; Lee et al., 1988, 1989; Laurence and Lang, 1988; Heagle et al., 1988; Hogsett et al., 1988; Rawlings et al., 1988; Cure et al., 1986; Lefohn and Runeckles, 1987; Lefohn et al., 1986, 1988, 1989; Krupa and Kickert, 1987. (Krupa and Nosal, 1989).

It is apparent from a number of the general reviews of this area that no one exposure index or dose statistic is best for all crop species. The rigorous statistical search for relationships between crop response and a large number and variety of ozone averaging periods or exposure indices generally points to the importance of peak concentrations and weighted, cumulative-exposure indices. However, in many cases, these relationships can be very complex and difficult to interpret from a standard setting basis (Runeckles, 1988; Lefohn et al., 1989).

Indices based on peak or weighted cumulative concentrations (number of hours equal to or above a minimum concentration) also are subject to limitations in that neither address the length of episodic events, intervals between episodes, the sensitivity of the target organism at the time of exposure or the amount of pollutant that enters the plant or canopy (Lefohn et al., 1988). Lee et al. (1988) attempted to overcome some of these limitations by the incorporation of a number of weighting systems in the analysis of some of the NCLAN data. They found that while no single index was best in relating ozone exposure to crop response, the indices that performed the best were those that cumulate the hourly concentrations over time, emphasize concentrations over 60 ppb and phenologically weight the exposure over the plant growth stage. Lefohn and Runeckles (1987) reviewed the extensive efforts which had been undertaken and attempted to relate these findings to the development of an ozone standard. They concluded that the form of an

ozone standard could be defined by exposure to a specific number of occurrences of the higher concentrations (multiple exposure standard). This could be expressed as "no more than X days per year on which the 1 hour average concentration of Y ppm ozone shall not be exceeded".

In response to the scientific debate which has emerged on the issue of exposure indices best suited to describe the biological response of plants to season long ozone exposures, the principals involved in the NCLAN program also examined some of their data to investigate the use of alternative dose statistics (Rawlings et al., 1988). Using three differential weightings of the hourly ozone concentrations (peak vs. non-peak, time of day of the exposure, and total hourly solar radiation) the authors concluded that the NCLAN dose statistic was a reasonable dose metric and that, contrary to the findings of Lefohn, Lee and others, the effective dose metric is not well described by the weighting of the data for peak concentrations. The performance of solar radiation as a weighting factor, representing a crude surrogate for crop phenology, suggested that the level of plant activity should be given greater weight or recognition in the development of dose indices.

To determine the potential impacts of ozone exposure on Ontario vegetation, and to relate these effects to Ontario's existing 1 hour air quality criterion of 80 ppb, the NCLAN exposure statistic (seasonal mean) was utilized in a recent re-analysis of ozone impacts on vegetation in Ontario (Pearson, 1989). While the limitations of the seasonal mean exposure statistic were recognized, the lack of any generally accepted, alternative index which could be utilized, resulted in a decision to proceed with the seasonal mean analysis of vegetation impacts. Because this analysis was not restricted to the utilization of field research results from the NCLAN program, and actually included many field studies conducted in Ontario for which on-site seasonal means were available, some of the biases which would result from equating crop responses to seasonal means without regard to the episodic nature of the hourly values comprising the mean were minimized.

Because of the uncertainties and difficulties associated with the seasonal mean as a vegetation exposure index, and the absence of a viable alternative at this time, it was decided that Ontario's 1 hour criterion would be utilized as a basis for standard setting and oxidant control. This required an examination of the relationship between these two variables under Ontario growing conditions. In this regard, statistical correlations and regression were run on the rural and combined (rural+urban) data set for the period from 1974-1988. Although the equation (second degree polynomial) which best fits the expanded data set (1974-1988) has changed slightly from the earlier findings (linear) in the 1984 crop loss assessment (Linzon et al., 1984), the results still confirm that in the fairly limited geographical area of Ontario, there is a relatively strong relationship between seasonal means and peak hourly values. Based on this relationship, seasonal means at or below about 35 ppb (the generally accepted minimal effect yield threshold as reported in the NCLAN work), would be achieved in rural areas of Ontario by the attainment of the 1 hour annual maximum ozone concentration of 80 ppb. However, 30 ppb is considered by Singh et al (1978) to be the yearly mean O_3 concentration in the northern

hemisphere. Therefore, the seasonal mean may have little if any real value.

It was emphasized in the Ontario assessment that achievement of this standard would not provide complete long-term protection to all sensitive crops, nor would it ensure protection against short term acute foliar injury under favourable climatic conditions. It would, however, result in significant improvements in terms of crop damage in Ontario, as rural monitoring sites in this area of Canada experience up to approximately 170 exceedances of the 80 ppb hourly criterion during worst case years for ozone.

Based on crop loss values which were determined from an analysis of the North American field research data and subsequently adjusted downwards to reflect uncertainties in agricultural, geographical, and experimental variables (Pearson, 1989), the value of increased productivity to 19 agricultural crops and ornamentals (turfgrass, Christmas trees and nursery stock) in Ontario was estimated at from 17 to 70 million dollars annually. The upper range represents approximately 4% of the total $ 1.9 billion in Ontario crop sales. Crops determined to be at greatest risk included: dry beans, potato, onion, hay, turnip, winter wheat, soybean, spinach, green bean, flue-cured tobacco, tomato and sweet corn. Marginally at risk crops included: cucumber, squash, pumpkin, melon, grape, burley tobacco and beet.

With the exception of Alberta, (Torn et al., 1987) and British Columbia (Runeckles, 1989; Rafiq, 1989), this type of multi-crop impact analysis has not been conducted in other areas of Canada. In Alberta, the analysis consisted of a review of the available literature for ozone response based on crops grown. This information was then compared with a limited amount of urban ozone monitoring data, and it was concluded that there were no identifiable risks to sensitive crops at this time. In British Columbia, processing peas and potato crops have been fumigated with ozone under field conditions to better define the exposure-response factors. In addition, a preliminary estimate of the crop loss due to ozone levels experienced in the lower Fraser Valley was undertaken in 1986. Although this study remains an internal document, Rafiq (1989) has indicated that losses were estimated at approximately 9 million dollars. This was based on a seasonal mean analysis, modified to reflect longer growing seasons and milder temperatures, compared to Ontario (Linzon et al., 1984).

In summation, although estimates have not been developed Canada-wide, the potential benefits from improving ambient air quality could be considerable. Total sales of crops and ornamentals in Canada amount to about 8.9 billion dollars annually. The contribution to this amount from the four provinces where foliar injury and/or yield impacts have been documented amounts to about 3 billion dollars annually (33% of total).

5.4.2.3 PAN

In 1960, another photochemical component of smog, peroxyacetyl nitrate (PAN), was identified as the cause of specific symptoms on Romaine lettuce and Swiss chard in

California (Stephens et al., 1961).

PAN is the principal member of a family of nitrogenous compounds that can be photochemically produced in polluted atmospheres. These compounds are of concern because of their extreme phytotoxicity at concentrations in the low parts per billion range. Severe foliar injuries and economic losses of susceptible crops in southern California have been documented, while injury symptoms on indigenous plant foliage have been reported in the Netherlands and Japan (Temple and Taylor, 1983).

On the basis of published reports of ambient air monitoring (Temple and Taylor, 1983), including sites in Alberta and Ontario (Peake et al., 1988; Corkum et al., 1986), it is apparent that PAN concentrations in Eastern North America and other locations in Europe are lower by a factor of five to ten compared to those in southern California.

PAN injury has been observed in the field (SW U.S.) following concentrations of 15 ppb for 4 hours (susceptible ornamentals) and 25-30 ppb on leafy crops. Based on these dose:response values, the potential for PAN injury to plants in Canada, where growing season concentrations average 1-2 ppb with occasional hourly peaks of up to 5-10 ppb, appears remote.

5.4.2.4 NITROGEN DIOXIDE

On the basis of the world literature which has recently been reviewed by Legge and Crowther (1987), nitrogen dioxide is the only oxide of nitrogen that is considered potentially phytotoxic within the range of common ambient air concentrations. Under high light intensities, about 6000 ppb for 2 hours are required to injure sensitive plant species such as bean, tomato and cucumber (Taylor and MacLean, 1970). Low light intensity increases sensitivity of plants, with injury developing after exposure to 2500 to 3000 ppb for 2 hours.

Nitrogen dioxide can injure the same plants as ozone and at the same time within the leaf tissue. Injury symptoms are, however, generally different. Long term exposure to nitrogen dioxide (250 ppb) can cause reductions in growth and yield (Legge and Crowther, 1987); however, this observation is based on a limited amount of research.

Since ambient nitrogen oxide concentrations in Canada rarely reach the short term injury threshold alone (Dann, 1989), recent research has focused on the effects of mixtures of two or more phytotoxic gases. Exposure mixtures involving nitrogen dioxide are discussed in greater detail in Section 5.4.2.5.

5.4.2.5 INTERACTIONS OF OZONE AND OTHER POLLUTANTS

Because plant life is rarely exposed to the influence of only one stress factor, extensive efforts were undertaken in the mid-1960's in which plants were subjected to combinations of ozone, sulphur dioxide, nitrogen dioxide and later, acid rain/fog. The results of these investigations have revealed a variety of interactions (synergistic, antagonistic and additive) on numerous plant processes. As effort in this area progressed, the focus shifted towards crop yield as the final determinant of interactive effects. In recent years a number of seasonal crop exposures under NCLAN type research conditions have been conducted (Takemoto et al., 1988; Kohut et al., 1987, 1988; Heggestad et al., 1986; Surano et al., 1987; Heagle et al., 1974, 1983; Temple et al., 1987; Kress et al., 1986; Reich and Amundson, 1984). In all cases, ozone in combination with sulphur dioxide and acidic rain/fog at exposure levels (but no dynamics) similar to those encountered under field conditions remote from specific point sources have not resulted in enhanced yield loss above the additive individual pollutant effects. Although the field studies have not covered all crops for which ozone exposure information is available, the body of evidence appears to rule out significant interactive effects involving ozone and these major regional pollutants. The potential impacts of pollutant interactions on Canadian crops are discussed in greater detail in Torn et al. (1987) and Pearson (1989).

5.4.3 EVIDENCE FOR EFFECTS ON TERRESTRIAL WILDLIFE

Acute effects of air pollution on terrestrial wildlife, such as mortality and reduced populations, have been reported in several instances (Schreiber and Newman, 1988). These effects appear to be largely local phenomena, however, and are not generally a consequence of long-range transport. Regional acidification of soils can influence the distribution of terrestrial biota, such as the salamander Plethodon cinereus, which is strongly reduced in abundance at soil pH's of 3.7 or less (Wyman and Hawksley-Lescault 1987). Other terrestrial wildlife can be affected in at least three ways by long-range airborne pollutants: through the loss of habitat, from changes in food availability, or as a result of a chronic accumulation of toxic pollutants in their tissues.

Forest decline has the potential to significantly impact forest wildlife. Large-scale losses of tree canopy are predicted to lead to a decreased abundance of birds that rely on the canopy for food and shelter (DesGranges, 1987). Other bird species that feed among shrub layers, on the ground or on dead trees may have increased habitat (DesGranges, 1987; Goriup, 1989), at least initially. Studies to test these predictions have been conducted in Quebec, in relation to the decline of sugar maples. Results have so far supported the predictions, with decreases in numbers of canopy birds and increases in shrub-gleaning species being found in dying vs healthy maple stands (DesGranges et al., 1987, Darveau et al., 1989). Similar patterns have also been detected using data from general bird surveys conducted in

southeastern Quebec (DesGranges, 1987).

Invertebrate communities that exist within forests and their soils can be altered by atmospheric pollutants including SO_2, mineral acids and ozone (Gilbert, 1971; Hagvar and Amundsen, 1981; Coleman and Jones, 1988). This in turn will modify the availability of food for birds and other wildlife, much in the same way that aquatic acidification has affected food supplies for waterbirds (McNicol et al., 1987). These relationships are currently being investigated for forest birds (Darveau et al., 1989, Gaulin and Mauffette, 1989).

In the Netherlands, where forest decline is a major problem, several species of forest birds are showing a rapid and alarming increase in the proportion of eggs with no shell, or with such a poor shell that embryos die of desiccation (Drent and Woldendorp, 1989). The authors hypothesized that a calcium deficiency is involved, brought on ultimately by acidic precipitation; acidic rain depresses the calcium:aluminum ratio in the soil, resulting in low calcium content of tree foliage and in caterpillars eaten by the birds. There have been no reports of such a phenomenon in Canadian forests at present, although low dietary calcium is known to be a concern for wildlife feeding in acidified environments (Scheuhammer, 1990).

Metals and other contaminants are transported long distances in the atmosphere, and can be accumulated by wildlife in remote areas. Cadmium is of particular concern because of its potential for bioaccumulation and its recognized toxic effects on various species of wildlife (Wren, 1987). Over 90% of cadmium in the air is derived from anthropogenic activity (Nriagu, 1980). A survey of Sphagnum fuscum moss from across Canada indicated elevated deposition of this metal in the vicinity of smelters (Glooschenko, 1989). In wildlife, elevated levels of cadmium are seen near sources of industrial pollutants (Kocan et al., 1980; Froslie et al., 1986; Scanlon et al., 1986), particularly near smelters (Sileo and Beyer, 1985; Crete et al., 1987; Reif et al., 1989). Recent evidence suggests that soil acidification may also result in higher cadmium burdens in wildlife, due to increased cadmium availability in soils and in terrestrial and aquatic plants eaten by wildlife (Parker, 1989; Hickie et al., 1989). In Ontario, cadmium levels in deer and moose tend to be higher in poorly buffered, acidic sensitive areas (Glooschenko et al., 1988). Further analysis of these data revealed that adult moose cadmium levels were significantly higher in a poorly buffered watershed than in a well buffered area. Aquatic forage plants preferred by moose also accumulated significantly more cadmium in the poorly buffered watershed (Hickie et al., 1989). Parker (1989) demonstrated similar patterns for deer feces and deer forage plants. In northern Europe, cadmium concentrations in wildlife reflect patterns of long-range atmospheric transport (Froslie et al., 1986; Steinnes, 1989).

Herbivorous mammals accumulate relatively high cadmium levels, mainly because of their consumption of plants that experience primary cadmium uptake (Frank, 1986). Moose, deer and caribou accumulate sufficiently high concentrations of cadmium in livers and kidneys that these organs have been declared unfit for human consumption in several provinces of Canada (Crete et al., 1987; 1989; Glooschenko

et al., 1988; Wotton and McEachern, 1988; Ecobichon et al., 1988).

To ascertain whether the aquatic feeding habits of moose are a factor, cadmium levels have been determined in three species of aquatic plants (Glooschencho et al., 1988). Manitoba Environment has also studied cadmium concentrations in carnivores, to determine if Cd is becoming concentrated in the food chain. Newfoundland Department of Environment and Lands is expanding their survey to include ptarmigan (Brazil, 1989).

Other contaminants (eg. organochlorines) that are transported in the atmosphere are also known to accumulate in wildlife in remote regions of the world, such as the Canadian arctic (e.g. Norstrom and Muir, 1988; Norstrom et al., 1988), and among offshore birds of the north Atlantic ocean (Pearce et al., 1989). Concentrations in remote areas are generally much lower than observed in locally contaminated sites. However, there is still reason for concern since some contaminants show no signs of decreasing over time, and the toxicological consequences of many are not well-known.

5.5 MITIGATIVE MEASURES

Because of the extent of the decline phenomenon, mainly in Quebec but also in the Maritime provinces and Ontario, demand for remedial measures is increasing. Unfortunately, as long as exact causes and mechanisms are not fully understood, such measures cannot really be defined. However, various studies of maple decline suggest a number of preventative measures which contribute to slowing down the progression of the phenomenon, at least in visually healthy or lightly affected stands. Furthermore, studies on the nutritive status of declining maple stands and subsequent fertilization trials have brought scientists to consider fertilization as an ameliorative measure.

5.5.1 PREVENTATIVE MEASURES

In a study of 243 semi-permanent plots in Quebec, Gagnon et al. (1985, 1986) have shown that high levels of decline in sugar maple stands are generally associated with extreme drainage conditions. The study also shows a negative correlation between basal area and level of decline. For example, in a healthy or slightly affected maple stand, the mean basal area is 32.7 m^2/ha compared with 15.3 m^2/ha when the decline level is more than 51%. Factorial analyses of 74 different variables (ecological, dendrometric, pathological, climatological, pedological, etc.) revealed that only 22 variables, mostly those related to pathology and soil characteristics, were significantly related to the level of decline. Typically, maple stands where the severity of decline is above 50%, grow on thin and rocky soils or on sites where the drainage is imperfect. Those highly-affected sugarbushes generally show common entomological, pathological, dendrometrical and pedological characteristics.

After studying 11,000 trees distributed in 110 research plots in Ontario, McLaughlin (1987) concluded that the decline phenomenon observed in the centre of the southern portion of the province in 1984 and 1985 was related to repeated tent caterpillar infestations in the mid-1970's. McLaughlin submitted that other factors, such as climate, diseases and the age of the trees, had also contributed to the decline, and that acidic rain was a potential additional stress on the forest ecosystem.

Even though Gagnon and McLaughlin disagree on the relative importance of the contribution of each factor to the decline phenomenon, the preventative measures which they suggest are similar (Gagnon et al., 1985; 1986; Gagon, 1988; McLaughlin, 1987; McLaughlin et al., 1989). Management priority should be given to maple stands which grow on mid-slope or bottom of the slope sites characterized by deeper and richer soils. It is also important to maintain the diversity of tree species and age in the stand, and to restrict thinning by removing only dead or dying trees. Any major opening of the canopy should be avoided. In order to limit the compaction of the soil, no animals or machinery should be allowed in the maple stand. Maple syrup producers should also avoid excessive tapping of their trees and fertilizers should not be applied unless a complete analysis of the nutrient status has been made.

5.5.2 AMELIORATIVE MEASURES

A recent paper by Hendershot and Jones (1989) has provided an excellent discussion of the possible causes of maple decline in Quebec. It is clear that although air pollution may be a factor in the decline, the heaviest mortality is found only in areas where severe insect attacks, seasonal climatic extremes (frost and drought), low soil nutrient status on marginal, high elevation sites and the highest deposition of wet SO_4^{2-} and NO_3^- have all occurred coincidentally in the 1980's. These authors indicate that although fertilization may not solve the problem, it may provide a protective function while the causal agents are being identified. There is considerable European experience with fertilizers which substantiates this claim (Zottl and Huttl, 1986; Huttl and Wisniewski, 1987). The European decline of high altitude silver fir (Abies alba, Mil.) and Norway spruce (Pices abies, L. Karst.) forests is closely linked, in most cases, to foliar nutrient deficiencies. Although the mechanisms for the development of the nutrient deficiencies are unclear, they may be due to accelerated leaching of nutrient cations from foliage or soils, to increased growth due to nitrogen via the atmosphere, or to decreased uptake by roots damaged by aluminum toxicity. The specific deficiency that occurs depends on the nature of the soil and parent material. Leaching may worsen the situation where the initial conditions were marginal. Magnesium is particularly deficient in many West German forest soils. Zottl and Huttl (1986) have demonstrated that calcium, magnesium and potassium deficiencies occur on acidic soils derived from granite. These deficiencies result from strong cation imbalances which have developed in the soil because of the acidification and leaching processes. Trees on some soils are now showing severe potassium and magnesium deficiencies in their foliage. On

some calcareous soils with near neutral pH, accelerated leaching can result in low magnesium and/or potassium levels, relative to the amount of calcium, leading to difficulty for trees in obtaining adequate amounts of magnesium or potassium. Zottl and Huttl (1986) and Huttl and Wisniewski (1987) found that by using fertilization, based on diagnostic foliar analyses, young conifer forests that were in a decline condition could be revived. Because of the success of these experiments and others, (Pearce, 1986) the German state of Baden-Wurttemburg is spending nearly $100 million for fertilization and replanting over the next five years. Fertilizer applications are being based on both foliage analyses and soil tests.

In Quebec, studies of the nutritive status of maple stands have shown a link between decline and nutrient imbalances in foliar tissues. Since 1984, deficiencies in potassium (< 0.55% dry wt.) and phosphorus (< 0.08% dry wt.) have been observed in conjunction with decline in the Appalachian region of Quebec (Bernier and Brazeau, 1988 a,b; Paré et Bernier, 1988; Hendershot and Lalande, 1988). In 1987, deficiencies in magnesium (0.03 to 0.09%) were also found to be associated with decline in the lower Laurentides (Bernier and Brazeau, 1988 c).

Even though the exact causes of these mineral deficiencies are not known (Bernier and Brazeau, 1988b), fertilization of declining maple stands in the Appalachian region has contributed to correcting such imbalances and to restoring the vigour of the trees (Hendershot et al., 1989). Although it is probably not a permanent solution, fertilization treatments applied four years ago still show positive effects on trees today (1988). Presumably, healthier trees will prove to be less vulnerable to decline.

In 1987, a study of 86 maple stands in the Beauce area, south of Quebec city, revealed that over 50 percent of the 860 sampled trees had foliar potassium concentrations below the critical level of 0.55%. In order to correct these deficiencies, nine fertilizer ratios were tested in 2 ha plots in 79 of the 86 stands initially sampled. These fertilizers, some of which contained lime, were applied at various rates ranging from 400 kg/ha to 700 kg/ha. Three months after fertilization, foliar concentrations of potassium, calcium and iron had increased by an average of 20% compared with pre-treatment values. In 1988, the foliar concentration of potassium had increased from pre-treatment 0.55% to 0.67%. In all, 87% of the treated plots showed improved nutrient balance (Ouimet and Fortin, 1989a,b). Best results were obtained with a potassium-dolomitic fertilizer on acidic soils and with a potassium based fertilizer (without dolomite) on soils which already contained high levels of calcium and magnesium. As part of the same study, an additional 256 maple stands were fertilized in 1989 on an experimental basis.

In Quebec, an operational fertilization program for sugar maple stands was recently implemented in light of these encouraging results. In 1989, approximately 6,200 ha were treated in the Beauce region. If results from the 1989 fertilization experiments are conclusive, it is hoped that the amount of fertilizers applied on an operational basis to declining maple stands in Quebec will be reduced to about 200 kg/ha.

In Ontario, a fertilization project coordinated by the Ministry of Environment was started in 1987 on four different sites. Based on this research, scientists hope to be able to make fertilization recommendations which would be adapted to Ontario conditions. In Nova Scotia, research is currently being done on the use of lime to counter the acidification of soils. This project, due for completion in 1992, also deals with the impacts of acidification of soils on the quality of maple products.

5.6 KNOWLEDGE GAPS AND RESEARCH NEEDS

The preceding sections in this report have described mechanisms by which air pollutants may adversely affect components of terrestrial ecosystems. They represent the general state of science and technology with respect to knowledge gained since the previous RMCC report published in 1986. It is clear that while significant progress has been achieved in certain subject areas, much uncertainty still remains, particularly in the interpretation of the role of air pollutants in current Canadian forest declines. This situation also applies to the European and American forest declines, and is not unique to Canada.

A considerable body of scientific literature exists which describes the interactions of single pollutants with a wide variety of tree, crop and other vascular and non-vascular species such as lichens and mosses. The bulk of this knowledge has been gleaned from laboratory experiments which have attempted in many instances to simulate growth environment as well as pollutant dose in as realistic a manner as possible. Considerable literature also exists detailing how soils respond to acidification. These studies have been completed both in the laboratory using soil profiles collected intact in the field and by irrigation of soils in the field. Notable advances have also been made towards our understanding of how forest canopies and soils modify incoming precipitation acidity and how this might in turn affect site quality and stream chemistry.

There are two forest declines in Canada, both of which began in the early 1980's. Despite air pollutants being implicated based on circumstantial evidence, the precise role(s) of air pollutants remains to be defined. This fact aptly reflects the current state of science and technology with respect to the impact of air pollutants on terrestrial systems. Large uncertainties remain in many areas. For instance, considerable controversy exists concerning the applicability of laboratory dose-response data to the natural or field situation. Is the response of a tree seedling to a given pollutant necessarily that which would be expected from a mature tree? Does the response of a branch to gaseous fumigation represent that of the whole tree? How can single tree data be transposed to the stand or ecosystem level? Will leaching of soil nutrients such as calcium and magnesium by acidic precipitation reduce site productivity over one or successive rotations? What are the most appropriate soil sensitivity indicators in terms of impacts on forest health and productivity? Can these criteria be related to atmospheric loading and biogeochemical cycling and utilized in the development of target and critical loads for Canadian soils? These are only some of the many questions which must be

addressed if we are to be able to more fully understand the consequences of air pollutant interaction with terrestrial systems. A more complete analysis of forestry LRTAP research needs has recently been assembled by Mayo (1987) as part of an Alberta program to assess potential impacts.

The terrestrial resources at risk from air pollutants in Canada are clearly immense, both in extent and economic value. Many knowledge gaps still exist which may inhibit both our interpretive and predictive capabilities. The more immediate, but by no means all of these are listed below. They are itemized under the two general questions posed by the RMCC to the terrestrial sector which form the basis for this document. Answers to the four specific questions also follow.

i **What levels of acidic deposition, ozone and associated pollutants can be tolerated without significantly affecting the terrestrial environment?**

In the case of forests, an unequivocal answer cannot be given at this time. Currently, some 15 million hectares of productive, commercial forests in Canada are receiving in excess of 20 kg/ha/yr of wet-deposited sulphate. In addition, forest areas along the Bay of Fundy coast in New Brunswick, the shoreline of Lake Superior in Ontario and at several high elevation sites in Quebec are now known to receive substantial additional amounts of cloud and fog deposited acidity, respectively. Despite the fact that experimental evidence exists for potential deleterious effects on sensitive vegetation processes at simulated rain pH's <4.2, no critical or threshold exposure levels have yet been developed for SO_4^{2-}, NO_3^- or any other acidity measure or component in terms of the Canadian forest ecosystem.

In the case of agricultural crops, scientific research conducted to date has not established any measurable or consistent yield response attributed to the direct effects of ambient levels of rain acidity. In spite of the fact that ambient rainfall acidity in many rural areas of Canada and the U.S. is currently at or in excess of experimental thresholds for foliar injury and reproductive effects, most crop research has now terminated. This situation is aggravated by the fact that most of the field work which was conducted was carried out on only a few crop species and even fewer cultivars of any one species.

In contrast, ozone induced foliar injury and, in some cases, yield reductions of several agricultural crops have been documented in many areas of Canada, including New Brunswick, southern Quebec, southern Ontario and parts of British Columbia. The majority of the controlled exposure-response field research conducted in the U.S. (NCLAN) is now the subject of extensive retrospective analysis in the search for a better exposure index with which to relate crop response. However, there is no question that at current ambient ozone levels, which regularly exceed the Canadian objective (82 ppb - 1 hour) in many regions of Canada, millions of dollars of crop damage (yield/quality/productivity losses) are being experienced each year. Ongoing efforts in the U.S. and Canada to reassess plant response to seasonal, ambient ozone profiles and to develop a more

representative exposure index should result in much improved estimates of crop impacts and areas of concern and the development of a more meaningful basis for an ambient air standard to protect terrestrial resources.

Considerable progress has been made in the mapping of forest and agricultural soil sensitivity to acidic deposition. Some 46% of the Canadian land surface mapped to date is considered highly sensitive, 21% moderately so and 23% less sensitive. While initial sensitivity maps are now available, no data exist to assign critical or threshold loadings to these potentially at risk soils. Linkages between soil acidification processes and potential impacts on Canadian forestry also have progressed. Although considerably more is now known about how acidity inputs affect soil nutrient availability, balance and cycling between terrestrial and aquatic systems, the relationship of these alterations as a direct or indirect causal or predisposing factor in forest decline remains unproven.

While the response of many plant species to individual pollutants under short term exposures in the laboratory is understood, the relevance of these data to field situations is not clear. Accordingly, most pollutant threshold limit:plant response based values that have been established can only be categorized as having potential effects under field situations. Field-oriented, exposure-response research is urgently needed to develop terrestrial based critical loading values for each of the LRTAP deposition type pollutants. This should include such areas as the comparative responses of juvenile versus mature foliage and seedlings versus whole trees. Secondly, exposure-response experiments should be conducted using mixtures of pollutants, both wet and dry. Thirdly, such experiments must include consideration of other stresses, for example, drought and winter desiccation, also known to play a role in current declines. Fourthly, research is required to address pollutant exposure as a predisposing factor to pathogens and insects as well as abiotic agents such as freezing and soil moisture stress.

Recent fertilizer field trials have indicated that remedial, mitigative measures may alleviate decline symptoms. These trials should be extended over a number of years and include a range of varying site conditions. Laboratory and field experiments (multi-element factorial) should be conducted in which the nutrient requirements of the species can be studied, as deficiencies in both macro- and micro-elements have been implicated in current declines.

In the case of field crops, where most research has been terminated due to a general lack of negative impacts at current ambient acidic wet deposition levels, it will be difficult to justify additional research funding. However, it should be emphasized that the existing data set has many shortcomings. For example, little is known on the vast majority of Canadian crops or cultivars, with what is known being limited to direct foliar effects without regard for interactions with other biotic and abiotic stresses to which crops are exposed. A discussion of some of the more specific research needs can be found in Torn et al. (1987). Although these were developed for Alberta, they reflect knowledge gaps and research needs for all Canadian crops.

In terms of gaseous pollutants, more research is needed on the response of crops and annuals to ambient levels of all LRTAP air pollutants, particularly ozone. Most existing crop response data from field exposures have been based on NCLAN style research which utilized a 7 or 12-hour seasonal mean index. However, retrospective analysis of this fairly large body of experimental data points to the need to re-evaluate exposure parameters to account for the cumulative and episodic nature of ozone throughout the growing season. This re-analysis will require additional field research under growing conditions typical of Canadian agricultural areas. In the case of agricultural crops, this research should focus initially on the most economically important crops as well as on crops for which comparative data exist from NCLAN style research. However, there are still many crops for which no or insufficient ozone response data exist and these too should be given priority if an accurate and complete assessment of ozone impact is to be gained. For instance the effect of ozone on the Canadian greenhouse industry has not yet been explored.

The review by Torn et al. (1987) also contains additional research needs that have relevance for all Canadian provinces.

In the case of tree species and forest stands, ozone response research will be even more difficult. As was the case with agronomic species, tree growth reduction can occur without visible symptoms; visible symptoms can occur without growth impacts and rankings of species susceptibility based on growth measures do not always correlate with those based on foliar symptoms. This latter finding is important because lack of growth reductions with a decreased photosynthetic area suggests compensations in carbon allocation and respiration. Another factor is the possibility that subtle growth reductions may be missed due to experimental variability and inadequate error control.

In his review of ozone impacts on trees, Pye (1988) summarized the factors which limit the extrapolation of short term response data to longer growth cycle conditions, to mature trees and subsequently to stand level yield. These difficulties, which in effect represent research needs, have been briefly summarized below:

Extrapolating from Short Term to Long Term Exposures

- as trees vary in their response to and recovery from ozone over time, the length and timing of the exposure and subsequent data collection can significantly alter the experimental outcome and conclusions

- as leaf phenology differs significantly between determinate and indeterminate tree species, the impact of ozone for a short duration will vary, depending on species type and exposure timing

- as conifers retain their foliage for periods well in excess of a year, the impact of a short duration ozone exposure during only part of this period is of limited value in terms of the full life-span of the foliage

Extrapolating from Seedlings to Mature Trees

- as the balance (ratio) between metabolically active (photosynthetic) and catabolically (respiration) dominant tissues decreases with age, the impact of ozone early in the life of a tree may not directly translate into equivalent effects later in the growth cycle

- as the micro environment in which a leaf grows affects its morphology, resulting in large differences within a mature canopy, the impact of ozone on a uniform set of seedling leaf types may not represent the complete range of foliar response within a mature canopy

- as water and nutritional transport and storage differ between young and old trees as cambial reserves increase, this may affect daily and seasonal patterns of stomatal conductance and influence ozone uptake and impact

Extrapolating from Individual Trees to Forest Stands

- as the distribution of tree sizes in a stand directly affects timber value, and as ozone impacts may directly or indirectly affect this distribution, stand volume and size distribution could be disproportionately altered; of key concern is whether stand processes will compensate for or amplify impacts on individual trees

- as ozone susceptibility of dominant and suppressed trees within a stand will vary depending on a host of phenotypic and genotypic factors, ozone impact assessment at the stand level requires a more comprehensive understanding of stand dynamics, microclimate, genetic composition and site quality than is provided from seedling level experimentation

Although some long-term data are available from soil column work, the buffering capacities and cycling properties of forest soils need to be further investigated. Some excellent process research already exists in this area. The potential contribution of mineral weathering to Canadian soils also needs to be understood if meaningful target loadings for sulphate and nitrate are to be calculated. The development of target and critical loadings at the forest ecosystem level is presently a long way in the future.

Large knowledge gaps exist in the deposition to and potential or real response of large proportions of the Canadian terrestrial mosaic, namely, wetlands, tundra and grasslands. The former have largely been used as biomonitors.

One prominent unknown with respect to forests is the source/sink question. That is,

what contribution does the forest itself naturally make to ambient ozone concentrations indirectly through natural emissions of precursors to tropospheric ozone, ie. volatile organic carbon. The value of soils and vegetation as a pollution sink also requires study.

Considerably more attention should be paid to atmospheric nitrogen and its potential effect on terrestrial systems. Target loadings for NO_3^- have been agreed to by the ECE but may not be substantiated by science. Will excess nitrate provide a fertilizing effect in the long-term, or will it upset nutrient cycling processes and overwintering success.

ii What improvements due to Canadian and/or United States emission reduction scenarios, can be expected in the terrestrial environment?

The answer to this question is predicated on a complete understanding of all components of Question # i. As this is not yet possible, this question can only be answered over time through continued long-term integrated monitoring of ecosystems intensively assessed before and after emission reduction programs were enforced. Presently, atmospheric model estimates are available only for wet SO_4^{2-} deposition. Taking Canadian and projected U.S. SO_2 control programs into account, SO_4^{2-} deposition to the Algoma (Ontario), Dorset (Ontario), Montmorency (Quebec) and Kejimkujik (Nova Scotia) LRTAP watershed sites is expected to decrease by 27%, 42%, 32% and 22% respectively by the year 2000, relative to that monitored in 1980.

It is not clear at this time whether loading reductions of this magnitude could be documented at the terrestrial surface in a short to medium time frame. Although some biochemically/nutritionally based early diagnosis tests have proven successful under certain pollution regimes and considerable advances are being made in the potential of remote sensing as a tool to detect forest stress, it is not clear whether these approaches will be sensitive enough to allow for adequate discrimination between air pollutants and other stress factors. Perhaps the longest record of atmospheric deposition and element flux through a terrestrial ecosystem is the data set from the Hubbard Brook Experimental Forest in New Hampshire. Sulphate deposition, flux through the hardwood forest and export from the watershed via forest streams has been monitored over 25 years (1963-1988). During that period, SO_2 emissions peaked in 1970 and have declined to the present. A long term decline in bulk precipitation SO_4^{2-} was reflected in streamwater chemistry. Streamwater pH, however, has not changed in response to sulphate loading, but rather due to a parallel decrease in basic cation deposition. The decrease in SO_4 deposition has not yet been correlated with any measurable change in ecosystem function.

The answer to this question is contingent on at least some of the knowledge gaps outlined above being addressed. Before remedial responses can be predicted, the dynamics and scope of the air pollution contribution to current declines and crop

loss must be more fully understood.

Another key factor is the absolute necessity of a comprehensive and integrated biological monitoring system that can serve to accurately establish soil, vegetation and wildlife baseline conditions, detect response over time and relate these alterations to changes in air quality. This should be in place for all ecosystems, regardless of their current state or health. If these monitoring systems can be integrated with current international initiatives, then all parties, regardless of their contribution to the problem will be in a better position to support appropriate abatement actions as required.

Progress has been achieved to date in certain systems to allow for estimation of how emission reductions, if of course such reductions result in changes in deposition, may affect element cycling within forest ecosystems at several sites. However, a considerable effort is required to test to what degree these site specific models can be extrapolated to a regional or forest type basis. Also, the magnitude of dry deposition remains unclear and much work is needed to clarify dry deposition rates to forest canopies and to separate the contribution of dry to wet components in throughfall chemistry.

Current estimates of ozone related crop loss in Ontario were possible because of an extensive pollutant monitoring program, including sites in rural areas. Large uncertainties remain, however, as to regional pollutant levels in most of Canada. Where data exist, they are often derived from monitors situated so far away, and in such inappropriate locations (urban centres) as to make data extrapolation to the agricultural, forest or wetland environment impossible. Thus, improvements in the condition of the terrestrial components of the Canadian environment can only be linked to emission reductions if adequate monitoring for the pollutants of concern can be implemented.

Effects of acidic rain and other pollutants such as heavy metals on terrestrial wildlife represent another area where research is badly needed. Wildlife impacts are going to be incremental, difficult to detect and will require early research into the magnitude and direction of population changes and species vigour expected to result from habitat alteration.

Some information is available on the effects of gaseous pollutants and acidification on forest soil biota but much work remains to identify sensitive invertebrate taxa and effects of their loss on the remaining biotic community. Additional data are required concerning the effects on quality of prey available to terrestrial animals, including changes in nutritional content and the accumulation of metals and possibly other contaminants over the long term.

iii What is the risk that acidic deposition, ozone and associated pollutants will damage the forest?

Forest Areas Likely to be Affected

Based on available air and precipitation monitoring results, there is a potential for air pollutant damage to Canadian forests. Air pollutants are currently directly or indirectly implicated in at least two declines, namely, the sugar maple decline in Quebec and Ontario and white birch decline in New Brunswick. Both these areas receive relatively high, episodic concentrations of ozone and moderate to high wet SO_4 loadings. In the case of sugar maple, soils have become deficient in several key nutrients since the 1960's. Although a definitive cause-effect relationship has not been demonstrated in either case, there is increasing scientific evidence that soil chemistry is an important factor.

At present, insufficient data exist to accurately forecast damage in other areas potentially at risk. While hardwoods are generally considered less sensitive to acidic deposition than conifers at the juvenile seedling stage, these data cannot be extrapolated to the mature tree, stand or ecosystem level. It is also important to note that acidic deposition has a direct effect on the growth and vigour of lichens and mosses. This has potentially serious implications as these species play a substantial ecological role in boreal, arctic and sub-arctic ecosystems, comprising a major winter food source for ungulates, controlling water and nutrient balance (cycling and storage) through mulching and improving the nitrogen balance of sub-arctic soils.

Ozone has been shown in the laboratory and under controlled ambient exposures to reduce growth and impair tree physiology and biochemistry at concentrations similar to those monitored in some forested areas of Canada. However, until such time as more monitoring is conducted in all forested areas of Canada, and better exposure-response relationships are developed for mature trees and forest stands, scientific quantification of the risk will not be possible.

As a result of these uncertainties, forests at most risk must, for the present, be considered those in the zones of greatest SO_4^{2-} and NO_3^- deposition and greatest frequency of exceedances of the 1 hour Canadian ozone objective. Of the 10 major forest regions in Canada, this would include the Coast Forest region (lower mainland B.C.), the Great Lakes St. Lawrence Forest region (southeastern Ontario and southern Quebec) and parts of the Acadian Forest region (Fundy coast, southwestern N.S.). Declines have been underway since the early 1980's in the two latter regions. Preliminary dendrochronological data for sugar maple from Ontario indicate that significant growth reductions have occurred in declining as well as outwardly healthy trees in regions most heavily impacted by LRTAP. Additionally, high elevation forests (Mt. Tremblant in Quebec) already under climatic and edaphic stress should be considered at risk.

Potential Magnitude of Effects

At current levels of air pollution, the magnitude of air pollutant damage to forests is impossible to predict with any degree of certainty. Except around strong point sources, air pollution can be viewed as just one of many stresses acting concurrently on the forest. Reductions in yield of a few percent may be impossible to detect using existing experimental and statistical methods and even more difficult to relate to one cause, such as air pollution, as the European and Scandanavian experiences have shown.

Extent of Economic Loss

The current extent of economic losses remains unknown at this time as the role of air pollutants as contributing stresses to existing declines and as additional stress co-factors in forests in general has not been established. This situation will not be resolved in the immediate future as it requires longer and more integrative experiments designed to elucidate exposure-response dynamics over the full life of a tree and forest stand.

Ambitious goals have been set for the forest industry, including a 40% increase in productivity by the year 2000. In an industry which contributed some 35 billion dollars in 1987 value of shipments, a 1 or 2% loss in yield on an annual basis could have far reaching consequences if it occurred in the more accessible and valuable forests.

iv **What role does acidic deposition, ozone and associated pollutants play in causing hardwood decline?**

Economic Consequences of Sugar Maple Decline

There are some 15,000 maple syrup producers in North America who rely on the sugarbush for all or part of their income. Quebec alone produces about 70% of the world's maple syrup. In the Appalachian region of Quebec, the frequency of trees sampled between 1984 and 1988 which were assigned to the severe decline class increased from 21 to 30%. Although individual producers have been forced to leave the industry due to high tree mortality, total industry revenues have not decreased due to the decline.

Mitigative Measures for Maple Decline

Precise mitigative and/or ameliorative methods can only be prescribed once the causes of the decline have been identified. Currently, sugar maple decline is

concluded to result from a complex of factors, natural and man-made. Heaviest maple mortality occurs where combinations of past severe insect attacks, seasonal climatic extremes, low soil nutrient status and high levels of ozone wet SO_4^{2-} and NO_3^- coincide.

Research is urgently needed to investigate the contributory roles of each of these stresses in combination with basic studies on maple stand management. In the meantime, while answers are being sought, fertilization appears as a promising technique for alleviating decline symptoms. This research may have additional benefits in that some of the more difficult causality relationships between soil chemistry and root health may be clarified in the factorial arrangements of the fertility component treatments now being evaluated.

v **What is the risk of acidic deposition, ozone and associated pollutants reducing agricultural production?**

Crops and Areas affected

As indicated in response to Question # 1, there has not been any measurable or consistent yield response attributed to the direct effects of ambient levels of rain acidity. Thus, although only a few crops, and even fewer crop cultivars have been evaluated under natural, exclusion conditions, the conclusion drawn is that currently, there are no measurable risks to agricultural crop production in Canada from current levels of rainfall acidity.

This conclusion will stand until such time as more research is conducted on other potentially sensitive crops and until better linkage is established to determine the relevance of the earlier controlled exposure,(ie. greenhouse/growth chamber) study results to field conditions. This body of experimental data certainly underscores the potential for adverse effects at acidity levels at or in excess of those frequently experienced in many areas of eastern Canada.

In the case of ozone, knowledge of causality with crop injury and yield reduction is significantly more advanced. However, with the exception of Ontario, Alberta and British Columbia, there have been no province-wide determinations of crops at risk in terms of yield reduction. There have, however, been several biomonitoring programs which have documented foliar injuries to a number of sensitive crops in New Brunswick (potato), Quebec (dry bean, soybean, tobacco), Ontario (dry bean, soybean, potato, tomato, onion, tobacco, cucumber, grapes, peanut) and British Columbia (pea, potato). Although none of these studies was designed to determine the exposure threshold for foliar injury, in all cases injuries were observed in locations where hourly ozone levels exceeded the Canadian objective at some point during the growing season.

Magnitude of the Effect and Economic Losses

Although there are no reported estimates of the magnitude of ozone impacts on large areas of Canada, the potential for significant economic benefits from improving ambient air quality are considerable. Total sales of crops and ornamentals in Canada amount to about 8.9 billion dollars annually. The contribution to this amount from the four provinces where foliar injury and/or yield impacts have been documented amounts to about 2.9 billion dollars annually (33%).

In Ontario, the estimates of increased crop and ornamental productivity from meeting the 1 hour objective (predicated on a seasonal mean:hourly ozone relationship) ranged up to 70 million dollars annually, representing up to approximately 4% of the total $ 1.9 billion in Ontario crop sales. Crops determined to be at greatest risk included dry beans, potato, onion, hay, turnip, winter wheat, soybean, spinach, green bean, flue-cured tobacco, tomato and sweet corn. Marginally at risk crops included cucumber, squash, pumpkin, melon, grape, burley tobacco and beet.

These estimates were based on a review of the North American literature dealing with season-long field exposures to ozone. This included but was not restricted to the NCLAN results. In many cases, studies had been conducted in Ontario under a wide range of ambient growing season ozone profiles utilizing chemical protectants.

In Alberta, a review of potential ozone impacts failed to find any identifiable risk based on a limited ozone monitoring database.

Open-air fumigations have been conducted in British Columbia. Although the findings have not yet been published, they do indicate that significant yield losses probably are occurring to sensitive crops in parts of this province. These data support a preliminary estimate of ozone impacts in the Fraser Valley which indicated annual damages could be about 9 million dollars. Again, the lack of rural ozone monitoring data has limited the application of these site specific research findings.

vi What is the risk of acidic deposition, ozone and associated pollutants affecting terrestrial wildlife?

The risk of air pollutants affecting wildlife is even more difficult to predict than the risk for forests or crops. Effects are likely to accrue indirectly over the long-term and result from a primary effect on habitat or on the quality of available food.

Defoliation due to tree decline is known to affect bird populations, while some macroinvertebrates and terrestrial amphibians are sensitive to soil pH. Acidification of the soil can also lead to a calcium deficiency for forest birds, at least in the Netherlands, where some forest birds are unable to lay eggs with normal eggshells. This phenomenon has not yet been studied in Canadian forests.

The food source of some ungulates, such as lichens eaten by caribou, and forage species preferred by moose and deer have also been shown to be efficient accumulators of atmospheric heavy metals. These species also accumulate metals (cadmium) in association with poorly buffered soils that are sensitive to acidification. This has resulted in demonstrated accumulation through the food chain and increases in tissue metal concentrations in grazing ungulates such as moose and deer.

The risk to terrestrial wildlife from air pollution on a larger, regional scale remains unknown.

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5.A APPENDIX

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